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# From deforestation to forest recovery: perspectives for the Amazon under the rule of the Brazilian Forest Code

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# Abstract

Continued tropical forests decline has drawn concerted attention by governments and distinct sectors of the civil society, which have responded with anti-deforestation policies and conservation strategies. Alongside conservation, large-scale forest restoration is crucial for counteracting the negative impacts of deforestation on socio-ecological processes. In this context, Brazil plays a pivotal role. Most of the Amazon, the largest continuous tropical forest in the world, lies within the Brazilian territory. Nearly 18% of the Brazilian Amazon forest cover was already lost, and land speculation, mining, and agricultural expansion continue to threaten the forest. Therefore, cutting back land use change emissions is a major pillar of Brazil's commitment to the Paris Agreement, which includes the plan to achieve zero net deforestation in the Amazon and restore 12Mha of forests countrywide by 2030. In this thesis I focused on the Brazilian Forest Code (BFC), the flagship environmental legislation governing land use in private lands of Brazil. In forestlands of the Amazon biome, the BFC requires the protection of 80% of the native vegetation as Legal Reserves (LRs). The latest version of the law, from 2012, also established the compliance conditions for past law offenders. Particularly, there are high expectations that the enforcement of the BFC will secure the protection of forests in LRs, and drive large-scale forest restoration. Therefore, my overall goal was to advance the knowledge about the potential of the BFC enforcement for the conservation of old- and regrowing forests in the Brazilian Amazon. Specifically, I (i) investigated the spatio-temporal patterns of net forest cover change for the influence area of the Cuiabá-Santarém highway, crossing the federal states of Pará and Mato Grosso in the Brazilian Amazon; (ii) evaluated the potential of the BFC enforcement for the protection of old and regrowing forests in the Brazilian Amazon, and estimated the contribution of regrowing forests for LRs demarcation; and (iii) applied a multicriteria analysis to map priority areas for large-scale forest restoration in private and public lands of Mato Grosso, contrasting the costs of restoration with the gains for biodiversity and carbon enhancement. Results show that the Cuiabá-Santarém focus region accumulated substantial deforestation, most of which on private lands, indicating a widespread non-compliance to the BFC. High net deforestation rates and decreasing prevalence of forest regrowth on deforested lands, indicates that this region is not near experiencing a turnaround from net forest losses to net forest gains. Hence, to promote forest expansion, it will be necessary to improve old- and regrowing forests governance. In this regard, results showed that if regrowing forests are included in LRs demarcation, over 6Mha of ongoing forest regeneration could be protected,

and one third of LRs deficits could be offset. Also, the future regulation of BFC compensation mechanisms will be key for determining the potential of the law for promoting restoration and old-growth forests protection additionality. Finally, a substantial variation in the spatial distribution of priority areas for forest restoration was identified across Mato Grosso, and for different scenarios. Private properties were key to enhance intensively deforested habitats, while restoration in public lands was more effective in reducing restoration costs and mitigating carbon. The findings of this thesis demonstrate the importance of detailed spatial information on land tenure and land use change in tropical areas, to support spatial planning, and address the potential of legal frameworks for promoting forest conservation and restoration. The estimates of legal protection of current regrowing forests have strong implications for Brazil's restoration targets. They call for an improved treatment of second-growth forests by federal and state legislations, and the creation of policy and mechanisms able to secure the protection of high-value regrowing forests.

# Zusammenfassung

Die anhaltende Entwaldung tropischer Regenwälder und die damit einhergehenden sozialen und ökologischen Folgen finden zunehmend Beachtung nationaler Regierungen und zivilgesellschaftlicher Akteure, die Initiativen zur Verringerung der Entwaldung und Strategien zum Schutz von Lebensräumen und Artenvielfalt entwickelt haben. Die Waldrestaurierung, d.h. die Wiederherstellung von Waldökosystemen, stellt hierbei, neben der Verringerung der Entwaldung, ein entscheidendes Ziel dar. Brasilien spielt in diesem Zusammenhang eine entscheidende Rolle. Der Großteildes Amazonas-Regenwaldes, der größte zusammenhängende tropische Regenwald der Welt, liegt auf brasilianischem Gebiet. Nahezu 18% des brasilianischen Regenwaldes sind bereits gerodet. Landspekulation, Bergbau, und Landwirtschaft stellen die stärkste Bedrohung für die Existenz des Regenwaldes dar. Im Rahmen des Pariser Abkommens hat sich Brasilien dazu verpflichtet die Netto-Entwaldung bis 2030 zu stoppen und 12 Millionen Hektar Waldökosysteme wiederherzustellen. Eine zentrale Rolle für die Umsetzung der Verpflichtung kommt dem brasilianischen Waldschutzgesetz (BFC) zu, der wichtigsten brasilianischen Umweltgesetzgebung, die die Rahmenbedingungen für die Landnutzung auf privatem Landbesitz regelt. Im brasilianischen Amazon verlangt das BFC den Schutz von 80% der natürlichen Vegetation, als sogenanntes Legal Reserves (LRs). In der neusten Gesetzesversion von 2012 wurde erstmals der Umgang mit denjenigen Landbesitzern festgelegt, die den Gesetzesvorgaben nicht entsprechen. Es wird erwartet, dass die Umsetzung des neuen BFCs auf der einen Seite den Schutz der Wälder in den LRs gewährleistet und auf der andern Seite, unter Mitwirkung der Landbesitzer, zu ein großflächige Waldrestaurierung führt. Vor diesem Hintergrund ist das Ziel dieser Dissertation die Potenziale des BFC für den Schutz der Ur- und den nachwachsenden Wäldern zu ermitteln. Im Speziellen habe ich in der vorliegenden Dissertation (i) die Raum-Zeitlichen Veränderungen der Waldflächen im Einflussbereich der Bundesstraße BR-163, zwischen Cuiabá und Santarém, analysiert; (ii) das Potenzial der BFC für den Schutz der des Regenwaldes und für die Waldrestaurierung bewertet; und (iii) prioritäre Gebiete für ein großflächige Waldrestaurierung, unter Einbezug von Kosten, Biodiversität und Kohlenstoffspeicherung, identifiziert. Die Ergebnisse zeigen, dass ein Großteil der massiven Entwaldung in der Region zwischen Cuiabá und Santarém auf privaten Grundstücken stattfand. Dies lässt auf eine weitverbreitete Nichteinhaltung des BFCs schließen. Hohe Netto-Entwaldungsraten und eine rückläufige Verbreitung nachwachsenden Waldes deutet

darauf hin, dass die Region weit von einer Trendwende von Netto-Waldverlust, zu Netto-Waldzuwachs entfernt ist. Um eine Ausbreitung der Wälder voranzutreiben, ist es daher notwendig, das Management der Ur- und nachwachsenden Wälder zu verbessern. Die Ergebnisse dieser Dissertation zeigen, dass mehr als 6 Millionen Hektar der derzeitigen Waldregeneration geschützt und ein Drittel der LR-Defizite ausgeglichen werden könnten, wenn die nachwachsenden Wälder in die Schutzzonen der LRs einbezogen werden. Die künftige Regulierung der BFC-Ausgleichsmechanismen wird einen entscheidenden Effekt auf die Waldrestaurierung und den Schutz der Urwälder haben. Die Analyse möglicher Regulierungs-Szenarien hat deutliche Variation zwischen prioritären Gebieten für die Waldrestaurierung in Mato Grosso gezeigt. Die Ergebnisse zeigen, dass die Waldrestaurierung auf privaten Grundstücken entscheidend für den Schutz von Biodiversität ist. Demgegenüber zeigt sich die Wiederherstellung von Waldökosystemen auf öffentlichem Land kostengünstiger und effektiver für die Kohlenstoffspeicherung. Die Ergebnisse demonstrieren die Relevanz detaillierter räumlicher Informationen zu Landbesitz und Landnutzungsänderungen, um die Auswirkungen von neuen rechtlichen Rahmenbedingungen für den Waldschutz und die Waldrestaurierung in tropischen Gebieten zu untersuchen. Die Schätzungen der derzeit nachwachsenden Waldflache, und dessen Schutzstatus, sind entscheidend um die nationalen Ziele der Waldrestaurierung zu erreichen. Die Ergebnisse verdeutlichen, dass ein besseres Management von nachwachsenden Waldökosystemen durch Bundes- und Landesgesetze notwendig ist, und neue Strategien und Mechanismen, die den Schutz nachwachsenden Wäldern sicherstellen, erarbeitet werden müssen.

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# General introduction

Extensive tropical deforestation has spurred environmental change, impacting landscapes, livelihoods, biodiversity, and ultimately contributing to climate change (MA 2005). However, in tropical agricultural systems, forest expansion is a common and often concomitant process to deforestation (Sloan 2008), and, in many regions, regrowing forests already comprehend the main share of forest cover (Blaser et al. 2011). *Second-growth* forests have an increasing importance due to their potential to mitigate the negative impacts of forest loss, and their sustainable management is strategic to maintain ecosystem services provision (Chazdon 2014).

In Brazil, large-scale forest restoration is an increasing priority, reinforced by international agreements. Current efforts are better coordinated at the Atlantic Forest biome (Brancalion et al. 2016b; Brasil 2017a; Crouzeilles et al. 2019b) while a lack in knowledge and strategies for forest restoration in other biomes persists. In the Brazilian Amazon biome, several studies have mapped the spatio-temporal patterns of forest regrowth and provided insights about forest succession trajectories, especially discussing its biophysical and ecological characteristics (Almeida 2008; Barlow et al. 2007; Berenguer et al. 2014; Zarin et al. 2005; Zarin et al. 2001). However, there is a lack of macro-scale studies investigating the implications of these patterns, for example, for environmental policies and governance, and to climate change (Aguiar et al. 2016; Carvalho et al. 2019a). In this thesis, I explore the policy relevance of forest regrowth in the Brazilian Amazon from, analyzing its potential contribution to rural landholders' compliance with the country's flagship environmental law, the Brazilian Forest Code (BFC). I then identify priority areas to promote forest restoration for achieving compliance with the BFC and restoration targets.

This introductory chapter is divided in two parts. In the first part, I provide a scientific and contextual background. I start with a brief overview of land use as a driver of environmental change, placing manmade land system changes in the broader process characterized by the beginning of a new geological epoch, the *Anthropocene*. I follow up with a deepened focus on tropical land use and land cover change and discuss pathways to promote large-scale forest restoration. Finally, I present my study area, the Brazilian Amazon, emphasizing historic processes of occupation, land use change, and environmental governance through the BFC. In the second part of the introduction, I present the thesis structure, the research questions, and objectives guiding the research.

# 1. Scientific background

## 1.1. Land use and land cover change as drivers of environmental change and as a pathway to sustainability transitions

The use of land to support human development has been responsible for the modification of natural ecosystems for millennia (Marlon et al. 2008; Pongratz et al. 2008). Circa 10,000 years ago, the beginning of agriculture and animal domestication supported population growth and the advent of complex societies across the globe (Ellis et al. 2013). Landscapes were transformed for food, settlement, and energy provision. But changes evolved more rapidly when western societies developed at the expenses of natural resources and labor exploitation, connecting people and places (or landscapes) across international markets (McNeill 2008; Thomas and Thompson 2013). In the last two centuries, urbanization, industrialization, and globalization installed a socio-ecological crisis, characterized by the pervasive degradation of ecosystems and uneven access to nature's resources by different societies (and groups within societies) (Dietz 2017; Newell 2008). This crisis is acknowledged by the different fields of knowledge and scientific disciplines, which propose distinct approaches to frame environmental problems, raise social awareness, or recommend solutions.

There is a growing consensus that the profound human footprint on the Earth system has led to the onset of a new geological epoch, coined as the *Anthropocene*[1] (Crutzen and Stoermer 2000; Lewis and Maslin 2015). Among the evidences of the Anthropocene (or that mankind has impacted natural systems on globally and on a geological time scale) are markers of changes in the cycles of matter and fluxes of energy beyond the Holocene normal variability[2], rising see levels, the destruction of the ozone layer, and the creation of anthropogenic geological sediments (Raupach and Canadell 2010; Steffen et al. 2011; Zalasiewicz et al. 2011).

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[1] The Anthropocene is widely acknowledged by academics and non-academics, but it is not yet an official geological time unit. Among the suggested dates for the onset of the Anthropocene are the "Great dying" – circa 1610 – (Koch et al. 2019; Lewis and Maslin 2015), the Industrial Revolution – circa 1850 – (Steffen et al. 2011) and the "Great Acceleration" period – circa 1964 – (Steffen et al. 2015a). All these periods are significant in history and transformed how humans interact with natural environments. According to Lewis and Maslin (2015) "the event or date chosen as the inception of the Anthropocene will affect the stories people construct about the ongoing development of human societies" p. 178. Hopefully, it will raise concern about human societies' limits to growth (Meadows et al. 1972).

[2] Global average atmospheric carbon dioxide ($CO_2$) records point an increase in $CO_2$ concentration from 280 ppm in pre-industrial times to 315 ppm in 1958 to more than 412 ppm in 2019 (NOAA 2019; Raupach and Canadell 2010). Perhaps the changes in the carbon cycle are the most emblematic within the Anthropocene, since $CO_2$ emissions – dominated by the combustion of fossil fuels – are recognized as the main driver of climate change.

*Land use and land cover change*[3] (LULCC) is one of the fundamental drivers of the Anthropocene (Ellis 2011; Zalasiewicz et al. 2012). Human activities have strongly impacted 70-75% of the ice-free terrestrial surface (IPBES 2019; IPCC 2019) creating *Anthropogenic Biomes* (Ellis and Ramankutty 2008). These areas cover forest and non-forest ecosystems, of which one third endured a complete conversion from a natural cover to cropland, grassland, planted forests, mining, settlement or infrastructure (Ellis et al. 2010). LULCC is directly connected to other important culprits of global environmental change, such as biodiversity decline (IPBES 2019; Maxwell et al. 2016; Oliver and Morecroft 2014) and climate change (IPCC 2019). Regarding climate change, the land sector contributes to almost one quarter (≈22%) of anthropogenic greenhouse gases (GHG) emissions (IPCC 2019), mostly associated to agricultural management (e.g., soil nutrients, livestock, rice cropping) and tropical deforestation (Smith et al. 2014). Specifically, deforestation leads to the decay and emission of carbon from biomass and soils, accounting for 9% of global carbon dioxide ($CO_2$) emissions, and destroys functioning carbon sinks, reducing terrestrial carbon uptake (Houghton et al. 2012; Smith et al. 2014).

The widespread conversion of ecosystems may lead to irreversible effects if ecological thresholds are exceeded (Davidson et al. 2012; Foley et al. 2005; Steffen et al. 2018). Uncertainties regarding such tipping points are high (Moore 2018) as LULCC is connected to other equally uncertain tipping elements (e.g., phosphorus and nitrogen cycles, biodiversity loss, and freshwater availability) (Lenton et al. 2008). Moreover, LULCC impacts on ecological functions differ across biomes and ecoregions, hence, conservation efforts have the role to identify and strategically allocate resources to priority areas for protection, which are heterogeneously distributed across the globe (Buchanan et al. 2011; Heck et al. 2018; Steffen et al. 2015b). Finally, LULCC has cumulative effects across scales (e.g., carbon losses due to forest conversion) and interacts with other Earth system processes, contributing to significant global environmental change (Lambin and Geist 2006; Rees 1995; Rockstrom et al. 2009).

[3] *Land cover* has been defined by Lambin and Geist (2006) as "the attributes of the Earth's land surface and immediate subsurface, including biota, soil, topography, surface and groundwater, and human (mainly built-up) structures" (p. 4). *Changes in land cover* may entail different gradients of alteration (driven by human or natural processes): from conversions (e.g., shift from forest cover to bare soil) to subtle modifications, measured across continuous values (e.g., decline in tree density), but insufficient to mark a shift between categories (Lambin and Geist 2006). Hence, land cover conversions will also depend on type of representation (discrete or continuous) and detail of the land cover assessment in place (Lambin et al. 2003). In turn, land use "has been defined as the purpose for which humans exploit the land cover" (Lambin and Geist 2006, p. 4) (e.g., agriculture, settlement, conservation), also including the way Earth's surface attributes are managed (e.g., management practices, inputs and gradients in use intensity).

The transition towards sustainable land systems is central to climate governance in the *Anthropocene* and to other pressing agendas (Kuyper et al. 2018). Sustainable land use is a cross-cutting theme to climate change mitigation and adaptation, food security, equity, and freshwater provision. The shift to sustainable land systems will require a strong reduction in rates and scale of undesired LULCC trajectories and stimulating socio-environmentally beneficial LULCC.

## 1.2. Forest loss and expansion in the tropics

Dense humid, seasonal and dry forests, and open woodlands circle the planet within tropical latitudes, originally covering around 10% of the global terrestrial surface (Figure I-1) (Lewis 2006). Tropical forests have paramount importance to the Earth system integrity (Snyder et al. 2004; Steffen et al. 2015b). They are home to more than half of the species on Earth, hosting hotspots of diversity and endemism (Gentry 1992; Jenkins et al. 2013; Kier et al. 2009; Pimm et al. 2014), being key to biodiversity protection (Gardner et al. 2009; Gibson et al. 2011). Tropical forests store a significant fraction of the terrestrial carbon, and are highly productive ecosystems, accounting for one third of the terrestrial gross and net primary productivity (Beer et al. 2010; Field et al. 1998). Tropical forests, and humid forests in particular, have an important role in climate regulation (e.g., rainfall and moisture circulation patterns, climate-cooling) and water cycling (e.g.,, infiltration, groundwater recharge, precipitation recycling) (Ellison et al. 2017) across scales (van der Ent et al. 2010). Unsurprisingly, tropical forests provide essential ecosystem services for people[4] and their protection and sustainable management should be taken as priority (Brockerhoff et al. 2017; MA 2005; Parrotta et al. 2012; Vira et al. 2015).

[4] The subsistence of over 25% of the world population (1.6 billion people) is partially or entirely dependent on ecosystem services provided by forests (FAO 2013). However, livelihoods dependency on forest services is complex to define and likely surpasses the one quarter estimate of the world population provided by the FAO (2013). Beneficiaries are not necessarily spatially connected to areas providing the benefits, but human (e.g., global trade of commodities) or natural (e.g., species migrations, moisture cycling) flows of people, biodiversity, materials and energy distribute ecosystem benefits to distant locations (Schröter et al. 2018; Serna-Chavez et al. 2014; Syrbe and Grunewald 2017).

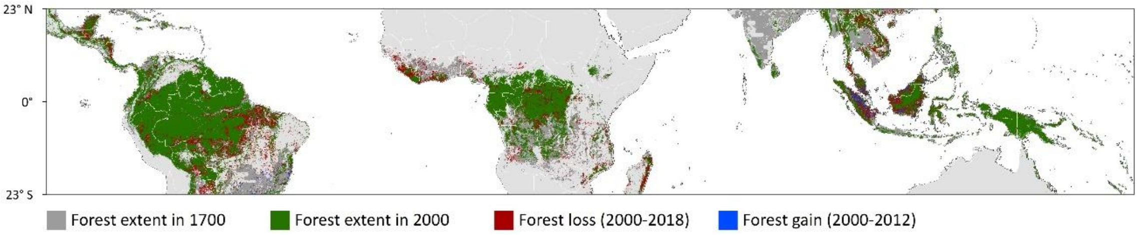



**Figure I-1. Tropical forests cover extent.** Original forest cover in 1700 (dark shade of grey) (Ellis et al. 2010), Forest extent in 2000 (green), hotspots of forest loss (red) and gain (blue) (Hansen et al. 2013), and non-tropical forest areas (light shade of grey).

However, the future of tropical forests is uncertain, threatened by climate change and persistent forest degradation and deforestation[5] (Aguiar et al. 2016; Coe et al. 2013; Hansen et al. 2013; Malhi et al. 2008). Tropical deforestation increased sharply in the 20$^{th}$ century, especially during the great acceleration period (approximately between 1945 and 2000) (Steffen et al. 2011). Tropical forests already lost 27.9% of their original cover (Ramankutty et al. 2008), and, in contrast to temperate forests, net deforestation is a persistent trend (Hansen et al. 2013). As pastures and croplands encroach, the increasing fragmentation makes forests vulnerable to disturbances (i.e., edge effects) (Broadbent et al. 2008; Murcia 1995). Fragmentation, combined with changes in canopy structure due to selective logging, exposes forests to wind and solar radiation, increasing the susceptibility to fires at forest edges and canopy gaps, triggering tree mortality and tree species impoverishment (Asner et al. 2004; McDowell et al. 2018; Okuda et al. 2003; Silvério et al. 2019). Concomitantly, the use of chemical defensives in agriculture impacts pollination, and habitat destruction combined with overhunting depauperizes the seed-disperser fauna (Laurance et al. 2002; Tabarelli et al. 2004). All these processes combined drive the loss of forest carbon stocks and weaken potential sinks from regrowing forests. Consequently, tropical forests are, in average, likely behaving as net sources of carbon (Pan et al. 2011).

Although currently outweighed by the carbon losses from tropical deforestation and degradation, tropical forest regrowth represents a large and expanding carbon sink (Pan et al. 2011; Rudel et al. 2016) and plays a strategic role for climate change mitigation and

[5] The term *deforestation* is commonly used to describe the human-driven removal of tree cover and vegetation for the economic exploitation of timber and (or) non-timber forest products and subsequentially making the land available for alternative uses, most often, agriculture (FAO 2007; Gibbs et al. 2010). Different concepts are used and depend on the policy context and monitoring systems' capabilities and objectives (e.g., distinguish anthropogenic from natural forest loss or between clearing of old-growth and second-growth forests) (INPE 2008; Sasaki and Putz 2009). For example, the FAO defines deforestation as *the conversion of forest to another land use or the long-term reduction of tree canopy cover below the 10% threshold* (FAO, 2007, p.5). Similarly, several studies have defined and measured deforestation in terms of extreme canopy cover loss (Defries et al. 2000; Hansen et al. 2013), whereas the term forest *degradation* is commonly used to describe a gradual change in forest structure, which may be associated with anthropogenic or natural disturbances (e.g., fires, selective logging, and biotic outbreaks) which may or may not lead to a full conversion of forest to other land covers(Pinheiro et al. 2016).

adaptation (Chazdon et al. 2016b). Forests naturally recovering from disturbance processes of varying intensity represent more than 50% of forest cover across the tropics, except for South America (Figure I-2) (Blaser et al. 2011; FAO 2010). Besides their potential to quickly recover carbon storage (Anderson-Teixeira et al. 2016; Martin et al. 2013), *second-growth* forests also provide habitat for less-demanding species, and improve connectivity between old-growth forest patches increasing the overall landscape value for biodiversity conservation (Arevalo-Sandi et al. 2018; Chazdon et al. 2009; Dunn 2004; Tambosi et al. 2014). They are important to support livelihoods through the provision of goods (e.g., timber, medicine, and food), soil conservation and nutrient cycling, and possible additional income from *Payments for Ecosystem Services* (PES) programs and ecotourism (Adams et al. 2016; Alves-Pinto et al. 2017; Xi et al. 2014). If taking place at large scales, where demand for assisted restoration exists, forest recovery creates economic benefits through the establishment of restoration markets (e.g., commercialization of seeds, saplings and equipment) and employment opportunities (Brancalion et al. 2013; Durigan et al. 2013; Erbaugh and Oldekop 2018).

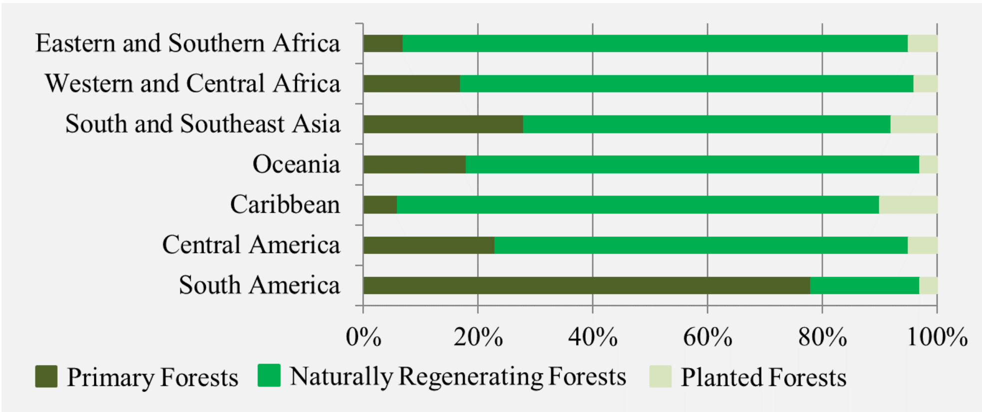



**Figure I-2. Broad categories of forests in tropical regions.** "Primary Forests" (old-growth forests) are "forests of native species, where there are no clearly visible indications of human activities and the ecological processes are not significantly disturbed" (FAO, 2010, p. 211) while "Naturally Regenerating Forests" present "clearly visible indications of human activities" (FAO, 2010, p. 211). "Planted Forests" are "predominantly composed of trees established through planting and/or deliberate seeding" (FAO, 2010, p. 212). Source: FAO (2010).

However, "forest gain is not the mirror-image opposite of forest loss" (Chazdon et al. 2016; p. 1). Forest regrowth is a far more heterogeneous process than deforestation, with implications for monitoring (Neeff et al. 2006) and for second-growth forests' value - for people and for the environment (Adams et al. 2016; Chazdon et al. 2009; Gardner et al. 2009). Regrowth trajectories are diverse, non-linear, and depend on several factors (Carvalho et al. 2019a; Chazdon et al. 2016a) (Figure I-3). The combination of the nature (i.e., anthropogenic or natural), intensity of the original forest disturbance (e.g., clear-cut, selective logging, fire), and the direct driver of forest reestablishment (e.g., silviculture, land abandonment, ecological restoration) produce very distinct second-growth forests (Chazdon 2014; Lambin and Meyfroidt 2010) (Figure I-3). For example, a long period of land use under intensive management may slow up or limit the reestablishment of ecosystem services (e.g., carbon sequestration) and forest traits (e.g., species richness and composition) by forest regrowth (Crouzeilles et al. 2016b; Wandelli and Fearnside 2015; Zarin et al. 2005). The landscape context also significantly impacts forest regrowth trajectories, since the proximity to contiguous old-growth forest patches favors regeneration (Crouzeilles et al. 2016a; Sloan et al. 2015). This spatial heterogeneity is important for planning and implementing forest restoration at the landscape level and needs to be considered by policymakers and practitioners.

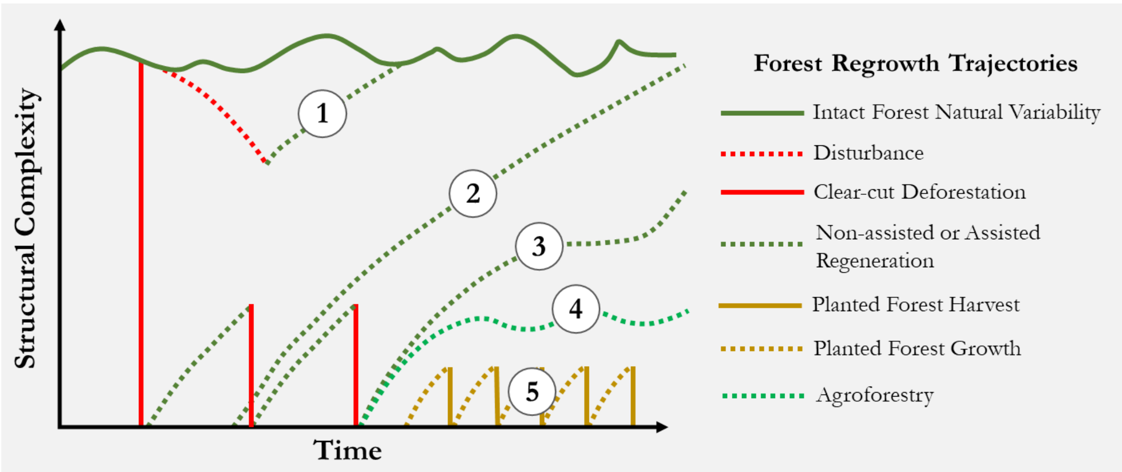



**Figure I-3. Temporal trajectories of regrowing forests displaying (forest structural complexity versus time).** In (1) forest recovers from a mild and gradual disturbance, quickly reestablishing the original complexity benchmark. In (2), following clear cut and a period of land use, forest regenerates and, after a longer period, reaches the original complexity benchmark. In (3) natural regeneration is arrested by consecutive clear-cut deforestation events and forest does not return to the original benchmark. In (4) agroforestry reestablishes some level of structural complexity that is kept stable through management. In (5) a commercial tree plantation exhibits simple forest structure and is cyclically harvested. Source: figure adapted from Chazdon et al. (2016a).

## 1.3. Governing forest transitions in the 21st century

Although not a silver bullet, tropical deforestation decrease, and large-scale forest recovery are crucial to achieve carbon emissions reduction and offsets (Bastin et al. 2019; Chazdon et al. 2016b; Lewis et al. 2019). The steady shift from net deforestation to net forest regrowth has been observed and extensively studied in several countries and regions and supported the formulation of a *Forest Transition* theory (Mather 1992; Mather and Needle 1998) comprising interrelated causal models of forest cover change pathways (Kull 2017; Meyfroidt et al. 2018; Rudel et al. 2005).

Forest transitions were first described based on forest cover trajectories of temperate countries (e.g., Scotland, France, Denmark, Japan), which endured a near-total forest depletion by the late 19th and mid-20th centuries, followed (not always immediately) by a consistent increase in forest cover (Mather 1992; Mather and Needle 1998). For these cases, industrialization and urbanization underpinned transitions in a cascade effect, characterized by workforce migration from rural areas to urban centers, leading to land abandonment in the countryside and consequent forest regrowth. This forest transition trajectory was termed as the "Economic Development" pathway (Rudel et al. 2005). In some cases, the demand for forestry products by growing urban populations and environmental problems (e.g., soil erosion and floods) also stimulated forest expansion (e.g., China's PES program "Grain for Green"), shaping a "Forest Scarcity" pathway to the forest transition (Mather 2007). Both pathways are linked to LULCC developments locally, but also in distant places (Lambin and Meyfroidt 2010). With reduced rural workforce, the demand for agricultural and forestry products is often met by processes of intensification (i.e., *land sparing*) and concentration of the production in the most suitable areas for agriculture (Lambin and Meyfroidt 2010; Rudel et al. 2009). However, the demand for land-based commodities is also frequently outsourced to other countries and regions. For example, the forest expansion observed in temperate countries partly occurred at the expenses of deforestation displacement to the tropics[6], an *indirect land use change* (iLUC) process (Meyfroidt et al. 2014) that defines the "Globalization and Displacement" pathway to forest transition (Meyfroidt and Lambin 2011; Meyfroidt et al. 2018). Finally, deforestation control and forest expansion can happen spontaneously or be the direct outcomes of state policies, to expand forestry or mitigate environmental impacts, a common driver of recent forest transitions (Rudel et al. 2019). This "State Forest Policy" pathway may also lead to iLUC (Meyfroidt and Lambin 2009). In this case, state policies at various government levels (national or sub-national) would make land a scarce asset, e.g., by forcing land set-asides in private lands or creating protected areas in public lands, leading to forest conservation and regrowth in spared areas but eventually to deforestation displacement across regions (Walker 2012) and countries (Pfaff and Walker 2010).

[6] However, this process also has been observed across the tropics (Meyfroidt and Lambin 2009).

The challenge for tropical countries with extensive old-growth forest cover is to accelerate their forest transitions, so that the turnaround point, from net forest loss to net gain, happens before the old-growth forests stocks reaches rock-bottom (Rudel et al. 2019). However, in many tropical regions, where deforestation frontiers are still expanding, old-growth forest loss and forest expansion are two sides of the same coin (Sloan 2008). This means that in deforestation frontiers, the demand for land and availability of land are very high, so that old-growth forests are continuously being converted, even while marginal lands are abandoned, and vegetation regrows. Deforestation exceeds forest regrowth, and second-growth vegetation also tends to be reincorporated into productive use, after fallow periods (Perz and Skole 2003). Therefore, fostering forest transitions in tropical countries is a governance challenge. It requires the elaboration and application of legal frameworks to enforce conservation of old growth-forests, as well as the identification of cost-effective opportunities for forest regrowth supported by local actors, sustaining environmental gains on the long term (Menz et al. 2013).

Environmental conservation discourses and values have increasingly been supporting forest transitions. A change in the public perception on the value of forests is taking place, backed by scientific evidence about the relevance of forest ecosystems for sensitive topics, and promoted by multilateral organizations and NGOs. This change appeals to governments to take on treaties and commit to deforestation reduction and forest restoration (Chazdon 2019; Chazdon et al. 2017; Kull 2017). In many cases, such global treaties, conventions, and mechanisms have been translated to national and sub-national commitments and policies, attempting to drive deforestation decline and forest recovery (Figure I-4) (Kull 2017; Meyfroidt et al. 2018). For example, the Bonn Challenge targets the restoration of 150Mha and 350Mha of degraded and deforested lands by 2020 and 2030, respectively. This target is supported by other initiatives, such as the Aichi Targets (2010) and the New York Declaration of Forests (2014) (Figure I-4), as well as by continental programs (e.g., 20x20 initiative in Latin America) (Chazdon et al. 2017). Several developing tropical countries have also ratified their Nationally Determined Contributions (NDC) at the UNFCCC Conference of the Parties (COP) in Paris (2015) which include land-based carbon mitigation pledges, many of which aligned with commitments made by signatories of the Bonn Challenge. This is the case of Brazil. The country committed to restore 12Mha of forests to the Bonn Challenge (Crouzeilles et al. 2019b) and as a carbon mitigation pledge, ratified by its NDC (Brasil 2015). To support this goal, a national vegetation restoration policy was proposed, largely reliant on the national legislation fulfillment (Brasil 2017a). At the regional scale, in the Atlantic Forest Biome, a coalition of social actors formed the Atlantic Forest Restoration Pact[7], which set their own commitment to the Bonn Challenge for the restoration of 1Mha of forests. This initiative promoted governance for forest restoration (e.g., knowledge creation and dissemination, interaction with policymakers, progress monitoring) and could be directly linked to the recovery of circa 0.7Mha of forests between 2011 and 2015 (Figure I-4) (Crouzeilles et al. 2019b). Hence, restoration initiatives led by coalitions between states and multiple social actors have been proven effective in driving large-scale forest expansion, and public policies will be essential to support forest transitions (Marcos-Martinez et al. 2018; Rudel et al. 2019).

As one moves from temperate to tropical countries or across transition pathways, it is important to realize that different forest transitions lead to forests of different quality and compared to the preexisting old-growth forests (Chazdon 2014; Chazdon et al. 2016a; Kull 2017). For example, the forest scarcity pathway is often associated with the plantation of native or exotic commercial tree species (e.g., Pinus, Eucalyptus, and Teak) (Rudel et al. 2005), that provide financial benefits, but perform poorly in the provision of other ecosystem services, being risky investments for long-term carbon storage (Wilson et al. 2017). Agroforests typically sequester less carbon and are less biodiverse in comparison to medium to advanced age second-growth forests naturally regrown (de Souza et al. 2016), but conserve soil, protect water resources, and are designed and managed to support livelihoods (Blinn et al. 2013; de Souza et al. 2016; Wilson et al. 2017). Natural regeneration provides goods, offers the opportunity to recover ecological traits on the long term, (typically) at lower financial costs, but its potential is context-specific and dependent on past land use duration and intensity (Figure I-3) (Chazdon and Guariguata 2016). Finally, unless people value such naturally regrown forests and conserve them, they will always be at risk of re-clearance if pressure for land resurges (Carvalho et al. 2019a).

[7] http://www.pactomataatlantica.org.br/

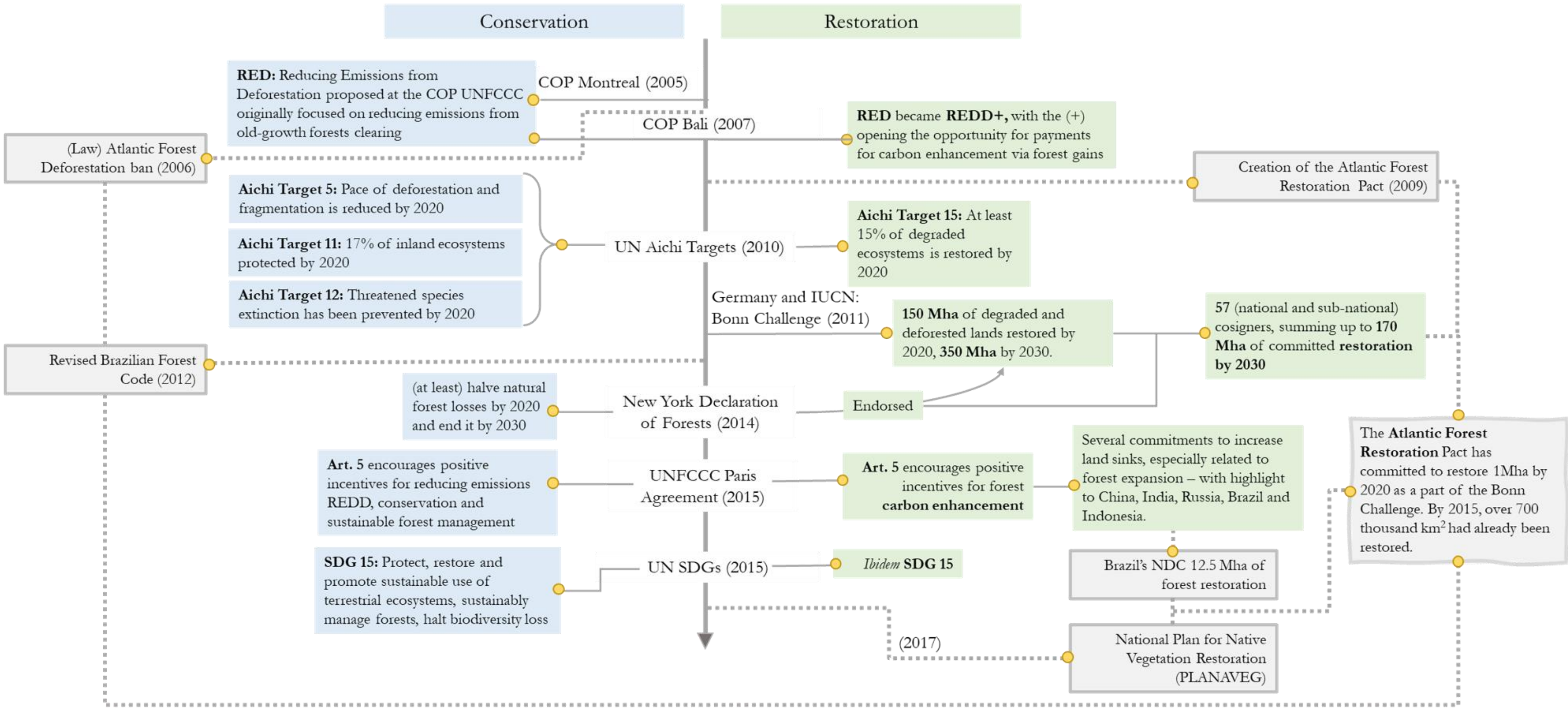



**Figure I-4. Main conservation and restoration mechanisms, treaties, targets and commitments made in the first two decades of the 21st century.** In blue and green conservation and restoration initiatives are highlighted, respectively. In grey I highlight different laws, mechanisms, targets and institutions that are relevant in Brazil, pointing their relation to international initiatives, taking FLR in the Atlantic Forest Biome as an example. Source: figure adapted from Chazdon et al. (2017)

Therefore, social actors promoting forest conservation and restoration with specific goals (e.g., climate mitigation and adaptation, downstream flooding prevention, or goods provision) must think of coherent pathways to achieve such targets. This necessarily involves understanding the drivers of deforestation and forest regrowth manifesting at multiple scales (Heck et al. 2018). Hence, effective conservation and restoration assessments and strategies must integrate knowledge about drivers at local (e.g., social actors agency, land tenure, and landscape features), regional (e.g., national or sub-national economic policies and legislation) and global scales (e.g., commodity trade and prices development, land-sensitive agendas and negotiations) (Heck et al. 2018; Meyfroidt et al. 2014).

In this context, researchers and practitioners argue that *Forest Landscape Restoration* (FLR) is a framework capable of promoting large-scale forest restoration (Sabogal et al. 2015). FLR aims to optimize the recovery of ecological functions at the landscape level, by identifying feasible restoration choices to support livelihoods and bring benefits for impacted social actors, considering the "broader pattern of land uses" (Sabogal et al. 2015, p. 4). FLR will identify restoration opportunities based on the institutional and geographical settings, "integrating multiple objectives and sustainable land uses to address the drivers and pressures that led to degradation in the first place" (Beatty et al. 2018, p.5).

Decision support tools have been applied to assist FLR planning, implementation, monitoring, and evaluation (Chazdon and Guariguata 2018). Such tools are used to guide the identification of suitable areas for distinct types of restoration strategies, minimize costs and maximize multiple objectives (Gourevitch et al. 2016; Nunes et al. 2017; Strassburg et al. 2019). Suitability criteria are context specific, but usually include information on the physical (Schulz and Schröder 2017; Tobon et al. 2017) and socio-economic feasibility of restoration (Gourevitch et al. 2016), the former often represented by economic return models of activities that compete for land with planned land restoration (Molin et al. 2018; Nunes et al. 2017). Benefits to be maximized are ideally aligned with the views of different stakeholders for regional development, and are commonly assessed by means of ecosystem services supply modeling (Gourevitch et al. 2016). Forest restoration costs can be high in degraded landscapes (Rodrigues et al. 2011) and are inversely proportional to the landscape resilience (Joly et al. 2014). In this regard, land use history and land cover indicators (e.g., percentage of remaining old-growth forest cover) are frequently used as proxies of landscape degradation and resilience, to assess the spatial distribution of restoration costs and the likelihood of restoration success (Crouzeilles et al. 2019a; Molin et al. 2018; Nunes et al. 2017).

During FLR planning, decision support tools integrate spatial and non-spatial layers of information, representing key objectives and constraints to forest restoration. These are often applied to multicriteria analyses, implemented using expert based evaluations about the relevance of the criteria (Tobon et al. 2017), spatial optimization algorithms (Thomson et al. 2009, Strassburg et al. 2019) to map priority areas for conservation and restoration interventions. Additionally, optimization models quantify potential trade-offs and synergies between concurrent land use objectives (Gourevitch et al. 2016). Scenario analysis are also often combined with participatory approaches and optimization algorithms to better understand the impacts of different climate, economic and land governance assumptions on prioritization outcomes (Zwiener et al. 2017). Finally, decision support tools can also guide the conciliation between social actors deciding for the best FLR intervention and location on the landscape.

# 2. The Brazilian Amazon: land use and environmental governance

## 2.1. Territorial occupation and land use policy

The Amazon Basin[8] covers nearly 7 million km$^2$ containing the largest swath of rainforest in the world (CDEA 2001). The extensive forest cover in the Amazon exists due to a combination of exceptional geographical features. The Andes Mountains, at the western edge of the basin, trap the moisture transported by eastern Atlantic winds, which precipitates over the rainforest (Salati and Vose 1984). Rainfall is intercepted, absorbed by plants and soil and further evapotranspirates. The vapor sent back to the atmosphere condenses, precipitating with the help of biogenic compounds released by the forest and oxidized by the sunlight (Poschl et al. 2010). This cycle is repeated several times, recycling moisture across the basin, and beyond its limits, and distributing rainfall over South America, which largely sustains rural and urban livelihoods south of the Amazon (Nobre 2014).

Even though the Amazon basin is shared among eight South American countries, over 65% of its extent lies within the Brazilian territory (CDEA 2001). The Brazilian Amazon biome covers 4.2 million km$^2$ (IBGE 2004) and ≈3.4 million km$^2$ of remaining old-growth forests (INPE 2014b) that sustain vital natural and human resources. According to Saatchi et al.

[8] The area drained by the Amazon basin is one of many ways to define the Amazon extent. For example, biogeographic sub-regions of the Amazon outspread the limits of the basin.

(2007), the Amazon forest stores 86±17 Pg of carbon, of which 47±9 Pg are in the Brazilian share of the basin, with a carbon sequestration potential of 0.49±0.18 Pg of carbon in an average year – in the absence of disturbance events (e.g., extreme droughts) (Phillips et al. 2009). However, by 2014, nearly 18% (0.76 million km²) of the original forest cover was already deforested (INPE 2014b), a striking consequence of an unsustainable economic development. Deforestation developed from east to west and south to north, concentrated along the *Arc of Deforestation*, majorly overlapping the states of Pará (PA), Mato Grosso (MT), Rondônia (RO) and Maranhão (MA) (Figure I-5).

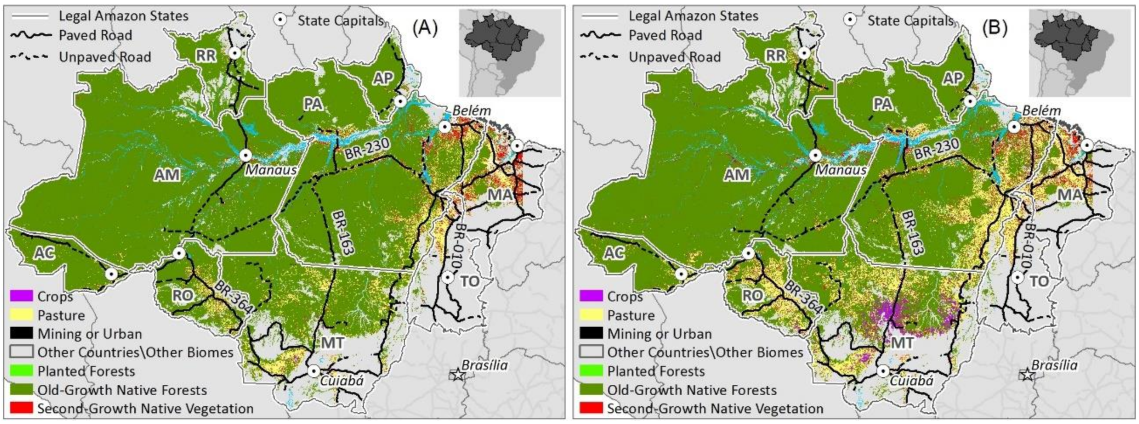



**Figure I-5. Land use and land cover in the Brazilian Amazon in (A) 1991 and (B) 2014.** The map highlights four main highways, the *Belém-Brasília* (BR-010), the *Brasília-Acre* or *Cuiabá-Porto Velho* (BR-364), the *Transamazônica* (BR-230) and the *Cuiabá-Santarém* (BR-163). States comprising the Legal Amazon: AC=Acre, AM=Amazonas, AP=Amapá, MA=Maranhão, MT= Mato Grosso, RO=Rondônia, RR=Roraima, TO=Tocantins. Source: INPE (2014c) and DNIT (2010).

Until the mid-20th century the Brazilian Amazon was a physically isolated and neglected region, and with a very small participation in the country's economic and political settings, apart from the short-lived rubber cycle during the late 19th and early 20th centuries. However, during the Second World War and particularly during the post-war period, the integration of the Amazon to the rest of Brazil became a geopolitical priority to Brazilian governments. During the 1950s, development agencies (SPVEA, *Superintendência do Plano para a Valorização Econômica da Amazônia* in Portuguese) and scientific research institutes were created (e.g., National Institute for Amazon Research, INPA) to support the "territorial occupation, socio-economic development and national integration" of the Amazon (Mougey 2018, p.386). As a result of such efforts, the first highway connecting the Amazon to central Brazil was inaugurated in 1960. The *Brasília-Belém* highway linked the national capital (Brasília) to Belém, located at the mouth of the Amazon river (Figure I-5). Overall, planning agencies regarded traditional extrativism as primitive, unable to scale-up socio-economic development and promote the integration of the Amazon to national and international markets, and from the onset sought to impose a technocratic view that praised commercial agriculture and cattle ranching as promising activities, organizing the territory accordingly (Mougey 2018). From the mid-1960s onwards, the military dictatorship (1964-1985) materialized this conception and intensified the Amazon's occupation (Carvalho 1991). New government institutions (SUDAM in 1966, *Superintendência para o desenvolvimento da Amazônia* in Portuguese) and strategic plans (PIN in 1970, *Plano de Integração Nacional* in Portuguese) were implemented, also aiming to attract corporate investments to drive economic development (Moran 1993). New highways were opened (Figure I-5), not only at the edges of the Amazon, but cutting through the forest (e.g., the *Transamazônica* and the *Cuiabá-Santarém,* inaugurated in 1973) (Becker 2005). In 1970, a new colonization agency (INCRA, *Instituto Nacional para Colonização e Reforma Agrária* in Portuguese) was founded and created several state sponsored settlements, mostly at the margins of the major highways opened during the period (Rausch 2014). A state-led agricultural research corporation was also founded in 1973 (EMBRAPA, *Empresa Agropecuária do Brasil* in Portuguese), advancing the knowledge to support large-scale agriculture and cattle ranching in Brazilian hinterlands of the Cerrado and the Amazon (Kaimowitz and Smith 2001).

The combination of a favorable institutional setting, mega-infrastructure projects, mining, access to credit, tax exemptions, and the promise of cheap land, attracted thousands of migrants from the northeast and south of Brazil, the latter settled mostly in private colonization projects in Mato Grosso (Arvor et al. 2016; Fearnside 2005; Jepson 2006). Despite of the heterogeneity of settlement frontiers (Rausch 2014), in many places landholdings evolved to latifundia covering the upland planes at the southern edge of the Amazon (Jepson 2006), often through processes of land grabbing (IPAM 2006; Paulino 2014). Consequently, deforestation spiked during the 1970s, violent land conflicts were crescent and threatened traditional livelihoods (Alston et al. 2000), and land speculation escalated, driving more deforestation (Fearnside 2002; Fearnside 2005). In the late 1980s, there was a reduction in the incentives for deforestation, due to economic recession and credit discontinuation, but deforestation soon picked-up again as agricultural production became increasingly coupled to international markets demands (Fearnside 2005). During the 1990s, internal (e.g., favorable currency exchange rates, internal market expansion, eradication of foot and mouth disease) and external influences (e.g., increasing demand for beef and animal fodder) favored the expansion of cattle ranching and soybean agroindustry, making deforestation in the Amazon more responsive to drivers of the globalized economy (Nepstad et al. 2006).

The Brazilian state encouraged colonization, with negligible environmental law enforcement. Quite the opposite, INCRA's colonization policies conflicted with legal land use restrictions in private properties, and deforestation frequently endorsed claims over public lands (Alston et al. 2000). The native vegetation protection requirements established by the Brazilian Forest Code (BFC), which is the piece of legislation governing forests on private lands in Brazil, were not respected, and farmlands accumulated forest deficits across time (more on the BFC in the next section) (Soares-Filho et al. 2014). However, around the mid-1980s, uncontrolled deforestation became a cause of concern by different civil society groups (Moran 1993), which led to governmental measures (e.g., anti-deforestation raids and financial incentives suspensions in 1991), albeit largely ineffective (Fearnside 2005).

A comprehensive program named PPCDAm (*Programa de Ação para Prevenção e Controle do Desmatamento na Amazônia* in Portuguese) was put in action in 2004, covering several policies to counteract an increasing trend in deforestation, changing forest governance in the Amazon (Figure I-6). Actions of the program included strategies for command and control (e.g., improved satellite monitoring coupled with field raids) (Borner et al. 2015), shaming (e.g., municipalities blacklist, public listing of environmental law offenders) (Cisneros et al. 2015), expansion of protected areas (Soares-Filho et al. 2010), and land tenure regularization (e.g., *Terra Legal* Program) (Brasil 2009; Oliveira 2013). In addition, increasing international pressure against deforestation in the Amazon motivated zero-deforestation commitments by soybean traders and leading meatpackers, who pledged to guarantee deforestation-free supply chains (Gibbs et al. 2016; Gibbs et al. 2015) (Figure I-6). Even though there are gaps in many of these policies (Gollnow et al. 2018; Imaflora 2016; Klingler et al. 2017), strategies put in place to stop deforestation, in combination with decreasing agricultural commodity prices in the late 2000s, contributed to an 84% reduction in deforestation rates between 2004 and 2012 (INPE 2014b) (Figure I-6).

The massive historical and future potential impacts of deforestation have justifiably made old-growth forests loss the most investigated LULCC process in the Amazon. However, forest regrowth has been receiving increased attention due to forest restoration commitments and its climate change mitigation and adaptation potential (Bustamante et al. 2019). Spontaneous forest regrowth is a common process in the Amazon, associated to the characteristics of productive agricultural systems (Perz and Skole 2003) and the dynamics of frontier expansion (Almeida 2008). Costa (2004) offers an interesting framework to understand the causes of forest regrowth in this region, categorizing forest resurgence as an

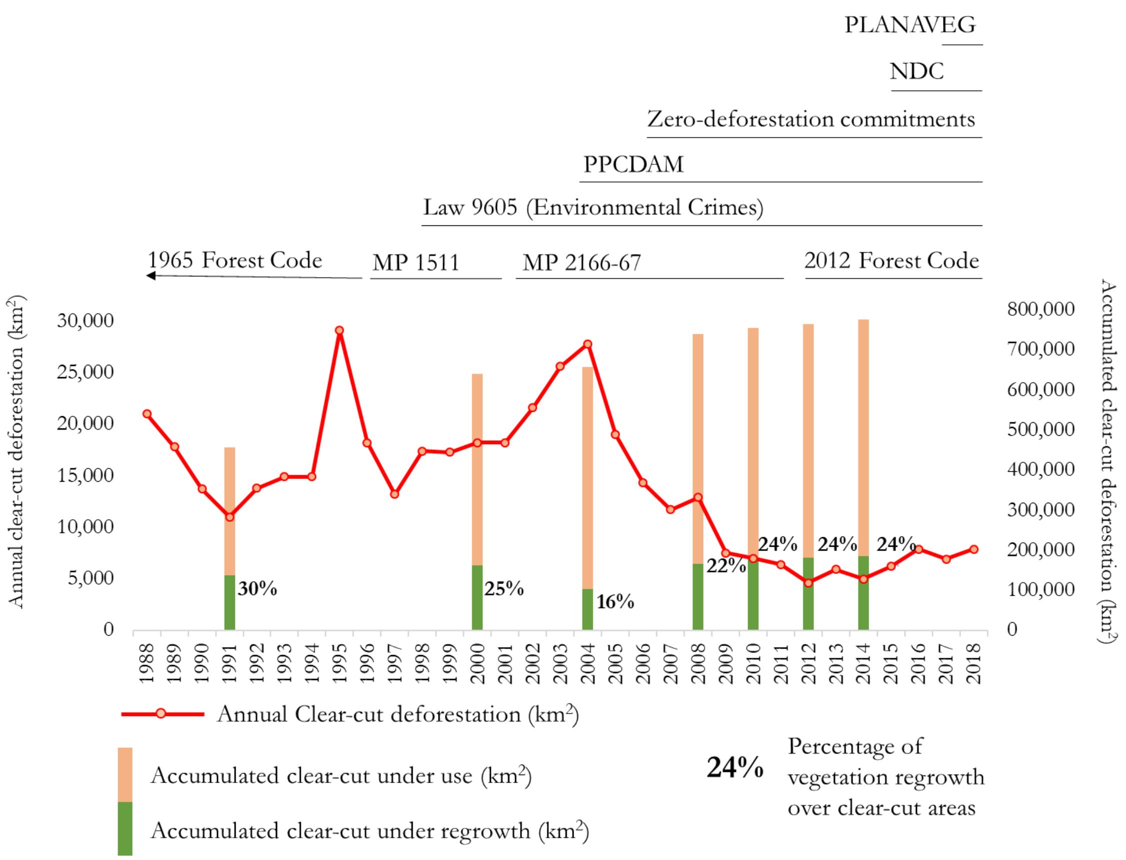



**Figure I-6. Annual old-growth forest clear-cut deforestation (red line) and accumulated clear-cut deforestation (bars).** Bars highlight the share (percentages) of accumulated deforestation covered by native vegetation regrowth (in green) in forest ecosystems. Vegetation regrowth data was only available for few years. Major environmental land use policies are annotated above the graph according to the date and period of duration. MP= Provisional Legal Measure. Source: INPE (2014b) and INPE (2014c).

endogenous component of shifting cultivation systems (Moran et al. 1996; Vieira et al. 1996) or as a by-product of the departure from agricultural systems supported by cyclic fallowing. In the latter case, forest regrowth would either be caused by land use intensification leading to land sparing (Costa 2007) or, more often, by degraded pastures abandonment following inadequate land management, which is one of the causes of the continued conversion of old-growth forests (Dias-Filho 2011; Nepstad et al. 1991; Uhl et al. 1988).

Forest regrowth in the Amazon is not a homogeneous process, neither spatially nor temporally (Soler et al. 2009). It is favored by traditional agricultural systems and agroforestry, while commercial agriculture typically suppresses regrowth (Perz and Skole 2003). Agrarian structure, represented by property size, has a negative effect on forest regrowth prevalence (Almeida et al. 2010; Perz and Skole 2003), but fallowing periods are shorter in small properties (D'Antona et al. 2006). On a regional level, the share of forests regrowing on previously cleared land is higher in new deforestation frontiers than in old frontiers, and forest succession decreases with frontier consolidation (Almeida et al. 2010; Soler et al. 2009). Finally, assessments confirm the continuous expansion of forest regrowth over cleared lands (Carreiras et al. 2006; INPE 2014c; Lucas et al. 2000; Neeff et al. 2006). However, the average age of regrowing forests has increased at a much slower pace (Neeff et al. 2006). Studies have shown that for most of the Brazilian Amazon, second-growth forests are young and short-lived (Almeida 2008; Carreiras et al. 2014; Müller et al. 2016b), which has important implications for forest restoration policy goals (Aguiar et al. 2016; Chazdon et al. 2016b).

An overlay between land categories and a land cover map shows that in 2014, forest regrowth was not homogeneously distributed across land tenure categories (Figure I-7). In many of the federal states of the Legal Amazon[9], most regrowing forests were in private lands and lands with unknown tenure (Figure I-7b). As a reference, remaining old-growth forests were mostly located in public lands designated for protection, sustainable use, and indigenous lands (Figure I-7a). Therefore, improved governance of regrowing forests in private properties should be regarded as an important component of forest restoration strategies in Brazil and in the Amazon region.

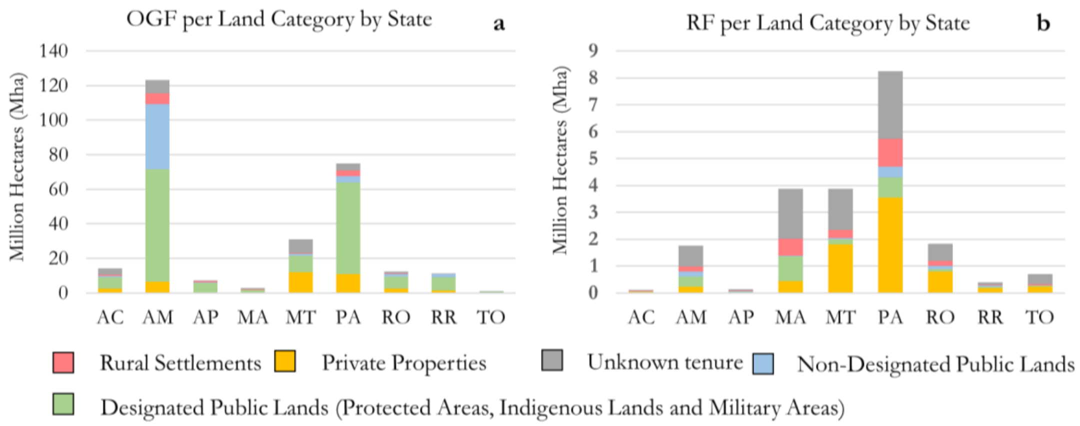



**Figure I-7. Distribution of forest stocks by land tenure category for federal states of the Brazilian Amazon.** (a) and (b) show the absolute (are in million hectares) distribution (%) of old-growth forests (OGF) and regrowing forests (RF), respectively. AC=Acre; AM=Amazonas; AP=Amapá; MA=Maranhão; MT=Mato Grosso; PA=Pará; RO=Rondônia; RR=Roraima; TO=Tocantins. OGF and RF extent refers to the year 2014. Source: INPE (2014b); INPE (2014c).

## 2.2. The Brazilian Forest Code and related legislation

The Native Vegetation Protection Law (*Lei de Proteção da Vegetação Nativa* in Portuguese), popularly known as the Brazilian Forest Code (BFC), establishes the rules for protection of forests and other types of native vegetation in rural properties (Brancalion et al. 2016a; Brasil

[9] A territorial planning region that was introduced during the 1950s to facilitate federal interventions promoting regional development, comprising the nine federal states that totally or partially overlap the Amazon Biome (Figure I-5).

2012). The current version of the law was enacted in 2012, but the history of the BFC dates to 1934. The 1934 BFC aimed to prevent uncontrolled conversion of forests in private properties, especially in environmentally fragile areas (e.g., springs, hillslopes, and riparian areas). However, it lacked clear guidance, and allowed up to 75% of clear-cut deforestation in private properties while the remainder 25% could be replaced by commercial forests with non-native tree species (Rajão et al. 2018). In 1965, a new version of the BFC was decreed and provided objective criteria to control land use in rural properties, stressing the need to protected native vegetation (Brancalion et al. 2016a; Brasil 1965). Specifically, it created two groups of required protected areas: The Permanent Protection Area (PPA or *Área de Preservação Permanente* in Portuguese) and the Legal Reserve (LR or *Reserva Legal* in Portuguese). The PPAs have a fixed location in the landholding determined by specific criteria. For example, the law determined that native vegetation on riparian corridors should be strictly protected, with buffer sizes that varied according to the river width. Other sensitive areas included in PPAs were slopes > 45º, hilltops, high elevations, plateau edges, estuarine vegetation, and buffers around pond margins and springs (Soares-Filho et al. 2014). The LRs are a required proportion of the landholdings in which the native vegetation cover should be maintained. The 1965 BFC established a regional differentiation in LR requirements between forest ecosystems in the Amazon, where 50% of landholdings should be set aside as LRs, and other biomes and consolidated areas, for which a 20% proportion was required (Brasil 1965; Rajão et al. 2018).

The 1965 BFC defined forests as a common good with juridical protection, restricting land use rights in private lands, and subjecting offenders to penalties (Stickler et al. 2013). However, incompliance to the rules was widespread (Soares-Filho et al. 2014). Rajão et al. (2018) argued that the lack of political will and unfamiliarity with the law contributed to this scenario. In addition, it was only in 1998 that environmental crimes were assigned tougher penalties (Law 9605; Figure I-6) and, nevertheless, enforcement remained weak (Brasil 1998; Nazareno 2012). Still, since the 1965 BFC enactment, several amendments were made as a response to concerns about environmental degradation. Most notably, a provisional measure (MP 1511) passed in 1996 in response to a spike in deforestation one year earlier, expanding the LR requirements from 50% to 80% of property areas in forestlands within the Amazon biome (Stickler et al. 2013) (Figure I-6). The 1996 MP was confirmed by an amendment in 2001 (MP 2166-67), that also stated that non-compliant farmers would have to abide to the BFC by restoring forest deficits.

The successive changes to the BFC (Figure I-6) left landholders uncertain of the regulations they should comply to (Stickler et al. 2013). As PPCDAm's command and control actions also targeted illegal deforestation in private lands, *ruralistas*, i.e., the agrarian oligarchies with political influence, lobbied for a new revision of the BFC with softer regularization requirements for historical offenders (Soares-Filho et al. 2014; Stickler et al. 2013). This political process led to the controversial version of the BFC passed in 2012 (Brasil 2012). On the upside, this version kept most conservation requirements for compliant properties, and set clear conditions for incompliant landholders to solve their forest deficits, giving juridical security for those willing to comply to continue producing. The 2012 BFC diversified the pathways to compliance. For example, it introduced a market mechanism allowing farmers to offset their deficits through the acquisition of forest certificates (CRA, *Cotas de Reserva Ambiental* in Portuguese) issued from rural properties with forest surplus (Soares-Filho et al. 2016). On the downside, the 2012 BFC forgave nearly 30Mha of LR and PPA deficits, reducing restoration requirements by incompliant landholders in 58% across the country (Soares-Filho et al. 2014).

Until recently, the magnitude and spatial distribution of the BFC forest deficits were largely unknown (Hirakuri 2003). However, as the BFC compliance became a pressing issue, and with the forthcoming state-legislations regulating the law implementation on the state level (Gasparinetti and Vilela 2018), policy-oriented research has helped to fill many knowledge gaps. Studies have estimated the BFC deficit (Sparovek et al. 2010), addressing the implications of the new BFC for restoration requirements (Soares-Filho et al. 2014), and ecosystem services provision (Aguiar et al. 2016; Alarcon et al. 2015; Garrastazú et al. 2015). They found that incompliance levels are higher in the Atlantic forest biome and along the arc of deforestation in the Brazilian Amazon (Soares-Filho et al. 2014). Following the 2012 BFC enactment, other studies also explored the conditions for the implementation of the new mechanisms for BFC compliance, particularly, the potential size of the CRA market (Soares-Filho et al. 2016), the role of land ownership uncertainties on the availability of forest certificates (Brito 2017), and the potential of distinct CRA market regulations for maximizing conservation additionality and socio-economic development (Freitas et al. 2017b). Other studies have identified areas where forest restoration for offsetting LR deficits is most cost-effective and feasible (Molin et al. 2018; Nunes et al. 2017; Strassburg et al. 2019).

These research advances were made possible by the increasing availability of high spatio-temporal resolution data on LULCC (Hansen et al. 2013; INPE 2014b), that allows monitoring forest stocks within properties and tracking the date of deforestation, to determine the eligibility to laxed BFC compliance conditions (Brasil 2012). Equally important was the creation of the rural environmental cadaster (CAR, *Cadastro Ambiental Rural* in Portuguese), a public land registry of rural properties boundaries, that allowed the verification of the compliance to BFC requirements for landholdings individually (Azevedo et al. 2017). The CAR enabled a refinement of BFC deficit estimates (Freitas et al. 2017b; Nunes et al. 2016) and research on the optimization of LRs placement, taking the landscape matrix into account (Kennedy et al. 2016b; Oakleaf et al. 2017).

## 3. The overarching goal of this thesis, research questions and objectives

Deforestation frontiers have progressively encroached into the Brazilian Amazon since the second half of the 20th century, largely disregarding the Brazilian Forest Code (BFC) However, the revision of the BFC in 2012 sparked expectations for a forthcoming law enforcement, which would be a natural and necessary development of ongoing policies targeting deforestation reduction (i.e., PPCDAm). Several studies followed, aiming to support good forest governance, and provided estimates of the BFC forest deficits in private lands, and analyzed the potential impacts of the law implementation under different scenarios (see section 2.2). In this thesis **I build upon this research with the with the aim to support the governance of regrowing forests in Brazil. I aim to better understand how spatio-temporal patterns of net forest cover change have developed to current levels of BFC compliance in private properties of the Brazilian Amazon biome. I evaluate the effect of future BFC regulations on the protection of old and regrowing forests. Moreover, I present a framework to support the spatial allocation of forest expansion for offsetting Legal Reserve deficits, providing estimates of monetary costs and potential outcomes for carbon storage and biodiversity**. The following specific research questions were addressed:

**Research Question I (Thesis Chapter II and Appendix Chapter):** What were the spatio-temporal patterns of net forest cover change in the Brazilian Amazon over the last decades for different tenure categories?

**Research Question II (Thesis Chapter III):** What are the outcomes of different Brazilian Forest Code implementation assumptions for the protection of current stocks of old- and regrowing forests?

**Research Question III (Thesis Chapter IV):** How do the costs and benefits of allocating forest restoration to private lands vary under different prioritization scenarios, and how do they compare with the outcomes of restoration in public lands?

The main objectives related to **Research Question I**:

Net forest cover change information (i.e., old-growth forest loss, forest regrowth and forest re-clearance) with high temporal and spatial resolution, is important to understand how processes of deforestation and forest regrowth evolved in response to distinct institutional settings, shaping the decline and recovery of ecosystem services. Specifically, investigating long term patterns of net forest cover change across land tenure regimes expands our understanding of landholders' decision-making rationale, providing valuable information for the design of policies aiming to curb-down deforestation and promote forest recovery.

Therefore, in **Chapter II** and the **Appendix Chapter** I address **Research Question I,** and investigate forest cover change using annual maps (between 1985 and 2012) covering most of the development of one important deforestation frontier in the Amazon: the influence area of the *Cuiabá-Santarém* (BR-163) highway, in Southern Brazilian Amazon (Figure I-8). I use carbon stocks storage and uptake as examples of ecosystem services provided by forests, to discuss the impacts of forest cover change across space, time and land tenure regimes.

*Objective (1)* Estimate net forest cover change and the associated carbon balance for different land tenure regimes in the influence area of the *Cuiabá-Santarém* highway (BR-163) between 1985-2012.

To address *Objective (1),* I apply the annual maps of old-growth deforestation, forest regrowth, and forest re-clearance to a carbon bookkeeping model to obtain an annual carbon balance from net forest cover change. I distinguish forest and carbon losses and gains for different land tenure regime categories (i.e., public designated lands, undesignated public lands, rural settlements, private properties registered with the CAR, and lands with unknown tenure). Finally, I interpret the historical trends in forest cover change for each land tenure regime, making associations with past socio-economical changes occurring at different scales (i.e., regional, national, and global), and discuss the potential of forest regrowth for mitigating carbon emissions from deforestation.

The main objectives related to **Research Question II**:

If the BFC is enforced, it will promote old-growth forests protection and drive forest restoration, even though estimates of the magnitude of its impacts vary. However, previous research has paid little attention to second-growth forests' role for the BFC implementation and for landholders compliance with the law, what is likely a consequence of the practically inexistent governance of regrowing forests (Vieira et al. 2014).

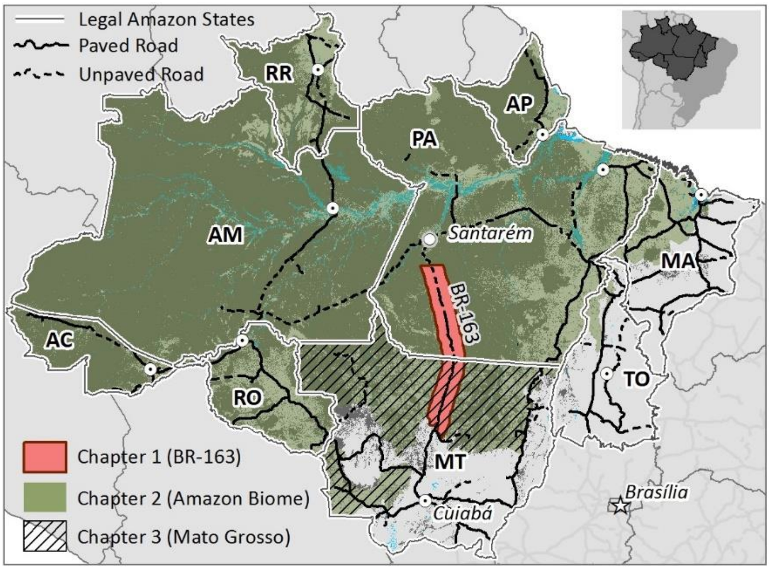



**Figure I-8. Overview of focus areas by each thesis chapter.** Chapter I focuses on the influence area of the *Cuiabá-Santarém* (BR-163) highway, a major deforestation hotspot. Chapter II focuses on forest ecosystems of the Amazon biome. Chapter III focuses on the portion of the state of Mato Grosso overlapping the Amazon biome. Source: INPE (2014c) and DNIT (2010).

In the Amazon, second-growth forests have historically been disregarded and seen as degraded lands, and their management has been entirely dependent on farmers' choices, with little interference from the public power at various government levels. The 2012 BFC also doesn't address second-growth forests properly, leaving for the states the task of regulating their eligibility to compose Legal Reserves (LRs) and issuing forest certificates (CRAs) (Vieira et al. 2014). In **Chapter III** I address this knowledge gap and provide estimates of BFC compliance distinguishing between the contributions of old- and regrowing forests. I focus on landholdings located in forest ecosystems within the Amazon biome, for all the states composing the Legal Amazon region (Figure I-8). Specifically, my objectives were:

*Objective (2)* Estimate the potential contribution of old- and regrowing forests to the demarcation of legal reserves, and to the availability of forest certificates (CRAs) for deficits compensation.

For this analysis, I use *(a)* spatially explicit land cover maps obtained from Brazilian deforestation and LULCC official monitoring programs for the year 2014 (INPE 2014b; INPE 2014c), resampled to a 100 meters spatial resolution, and *(b)* individual property boundaries obtained from the private lands registry (CAR). I use this data to quantify the extent of old- and regrowing forests in each rural property. I apply this information to a rule-based model working at the property level, that incorporates the BFC requirements to estimate: the extent of LRs and of forest areas eligible to issue CRAs depending on the BFC regulations, tracking the contributions from old- and regrowing forests, and LRs deficit area. This model also calculates the demand for future restoration, which may be interpreted as the demand for CRAs when the market-based mechanism for LR deficits offsetting is implemented. Finally, I present regulatory setups on the use of CRAs for LRs deficits compensation, to discuss the impacts of different BFC implementation rules on attainable deficit offsets and on the increase of additional forest protection.

*Objective (3)* Quantify the impacts of the BFC regulatory setups on the protection of forest carbon stocks

To address *Objective (3)* I use a spatially explicit dataset of above and below ground forest biomass to derive estimates of forest carbon stocks in private properties. I estimate the amount of forest carbon stocks that would be protected or be left unprotected if each regulatory setup from *Objective (2)* was implemented. I then contrast these estimates with those of forgone carbon uptake amounts due to LR deficits offsets made possible through CRA transactions, according to each regulatory setup.

The main objectives related to **Research Question III**:

For several countries that have pledged to engage in large-scale forest restoration, the identification of prime areas for forest expansion has become a land use planning priority (Gourevitch et al. 2016; Tobon et al. 2017). In the Brazilian context, previous studies have proposed modeling frameworks targeting an efficient allocation of forest restoration to offset LR deficits (Kennedy et al. 2016a; Kennedy et al. 2016b; Oakleaf et al. 2017). In **Chapter IV**, I present a prioritization model to allocate forest restoration across rural private properties with LR deficits. Previous research has recommended the restoration of large expanses of forest at the southeastern edge of the Amazon to maintain the regional hydrological cycle, prevent a forest dieback, and as an adaptation strategy to the effects of climate change (Lapola et al. 2018; Lovejoy and Nobre 2018). Therefore, I focus on Mato Grosso (Figure I-8), the Legal Amazon state with the large extent of LR deficits and address the state's commitment to allocate 1.9 Mha of forest restoration to properties with LR deficits by 2030. Specifically, my objectives were:

*Objective (4)* Propose a prioritization model capable of addressing multiple objectives of forest restoration at the landscape level

To address *Objective (4)* I propose a landscape modeling approach running with a compilation of spatially explicit datasets with a 100 meters' spatial resolution. This approach combines four objectives of forest restoration (a) habitat suitability for multiple species, (b) carbon enhancement, (c) likelihood of forest restoration, and (d) minimization of opportunity costs. The model uses a prioritization algorithm to select properties with LR deficits (as identified by *Objective 2*), and to allocate forests within landholdings, in a cost-effective way. I also test prioritization outcomes for different scenarios changing the weights of restoration objectives.

*Objective (5)* Comparing the costs and benefits of forest restoration in private and public lands in the state of Mato Grosso

To address *Objective (5),* I use the modeling framework proposed in *Objective (4)* to compare the costs and benefits of allocating restoration to lands under different tenure regimes in Mato Grosso, again using different scenarios changing the weights of restoration objectives.

## 3.1. Structure of the thesis

This thesis consists of five chapters. This introduction (Chapter I) provided a scientific background to the topic, contextualized the study area, presented the motivation, questions, and objectives driving the research. The introduction is followed by three core chapters (Chapters II-IV) each one addressing the research questions and objectives outlined by the previous section. Chapter V provides a synthesis of this thesis, summarizing the main research findings of Chapters II-IV, drawing conclusions and discussing its main implications. An appendix supplements Chapter II, providing the methodology for the carbon balance quantification and partly addressing Research Question I. The core Chapters II-IV and the Appendix Chapter comprehend four scientific articles prepared for publication within international peer reviewed journals - two of them are published and two are under review.

**Chapter II:** Land tenure diversity and forest cover change in a deforestation frontier in Brazilian Amazon. Hissa, L.B.V.; Gollnow, F.; Muller, H.; Lakes, T. (Manuscript under review, Land)

**Chapter III:** Regrowing forests contribution to law compliance and carbon storage in private properties of the Brazilian Amazon. Hissa, L.B.V.; Aguiar, A.P.D.; Camargo, R.R.; Lima, L.S.; Gollnow, F.; Lakes, T. Land Use Policy, volume 88, pages 1-10, November 2019

**Chapter IV**: Enhancing synergies for effective large-scale forest restoration in Mato Grosso, Southern Brazilian Amazon. Hissa, L.B.V.; Garcia-Marquez, J.R.; Gollnow, F.; Lakes, T. (Manuscript under review, Elementa: Science of the Anthropocene)

**Appendix Chapter:** Historical carbon fluxes in the expanding deforestation frontier of Southern Brazilian Amazonia (1985-2012). Hissa, L.B.V.; Muller, H.; Aguiar, A.P.D.; Hostert, P.; Lakes, T. Regional Environmental Change, volume 18, pages 77-89, November 2016.

# Chapter II

# Land tenure diversity and forest cover change in a deforestation frontier in Brazilian Amazonia

Letícia de Barros Viana Hissa, Florian Gollnow, Hannes Müller, and Tobia Lakes
*Land (under review)*

Submitted: 23 October 2019

## Abstract

In the Brazilian Amazon, vast tracts of old-growth forests are being cleared at varying rates in space and time. Differences in land tenure are key for understanding land management strategies affecting deforestation, forest regrowth and re-clearance (i.e.,, forest cover change and hereafter referred to as FCC). However, while extensive research has investigated the association between land tenure and old-growth forest clearing, less is known about how different ownership regimes influence forest regrowth and re-clearing. In this paper, we provide insights about net FCC and associated carbon outcomes across different land tenure regimes, for one of the most dynamic hotspots of forest loss in Southern Brazilian Amazon. We covered nearly three decades (1985-2012) of FCC taking place at the influence area of the BR-163 (*Cuiabá-Santarém*) highway, in Mato Grosso (MT) and Pará (PA). We accounted for five tenure categories (undesignated public lands, designated public lands, private properties, rural settlements and lands with unknown tenure. Our results indicated different FCC for tenure systems. Old-growth forests stocks were severely depleted between 1985 and 2012, with private properties undergoing the largest absolute losses in both states. Private properties showed a marked reduction in deforestation during the mid-2000s, likely associated with macroeconomic changes as well as mixed public and private anti-deforestation policies. In the state of PA, large expanses of public forests did not display similar decreases in deforestation, suggesting an unresponsiveness of lands with insecure tenure to anti-deforestation policies. Land designation to sustainable uses, combined with law enforcement could help to prevent additional deforestation in these areas. All tenure categories – including designated public forests, largely composed of conservation units – behaved as carbon sources rather than sinks. In none of the tenure categories forest regrowth appears to be associated with a resurge in forest cover but is rather a temporary component of the agricultural productive systems.

# 1. Introduction

Tropical forests around the world are being cleared at alarming rates (Hansen et al. 2013) with negative implications for the climate, biodiversity, and traditional livelihoods (Norris 2016; Rosa et al. 2016; Young et al. 2016). Recent research has highlighted the importance of forests and forest recovery as a natural and effective solution to support climate change mitigation and adaptation (Bastin et al. 2019; Griscom et al. 2017). This renders forest conservation and restoration as fundamental components of climate change mitigation plans of many tropical countries, ratified as Nationally Determined Contributions (NDCs) by the Paris Agreement (UNFCCC 2015). Such commitments are intertwined with other initiatives at global (i.e., Bonn Challenge, New York Declaration on Forests) or regional (i.e., 20x20 Initiative, REDD+) levels, on the hope to fast-forward forest conservation and expansion in the tropics (Chazdon et al. 2017; IUCN 2017).

Forest regrowth is a common process following deforestation and land use change in tropical agricultural systems, such as shifting cultivation (van Vliet et al. 2013; van Vliet et al. 2012). Occasionally, forest regrowth may also be associated with agricultural intensification, land degradation, land abandonment and rural out-migration (Costa 2016; Kuemmerle et al. 2011; Meyfroidt et al. 2010). However, even though regrowing forests are expanding rapidly - representing more than 50% of forest cover in the tropics (Blaser et al. 2011) -, in most cases, their short-life limits their potential to land-lock carbon on the long-term (Aguiar et al. 2016) and their value for biodiversity conservation (Barlow et al. 2007). Regrowing tropical forests rarely persist on the landscape for more than 5 years and are highly susceptible to forest fires due to their typical patch geometry, size and influence of poor land management (Alencar et al. 2015; Aragao and Shimabukuro 2010; Müller et al. 2016b; Schwartz et al. 2017).

Land tenure is one important predictor of forest cover change (FCC) in the tropics. Land tenure systems mediate the relationship between social actors amongst themselves and towards land and its resources, defining access, use, withdrawal, and alienation rights and duties (Corbera et al. 2011; Schlager and Ostrom 1992). These differences cause diverse land management strategies by landholders leading to distinct FCC outcomes (Paneque-Gálvez et al. 2013; Perz et al. 2017). For example, in Brazil, rapid deforestation was an outcome of the establishment of agrarian reform rural settlements, as settlers were expected to clear and keep the land under productive use to secure tenure (Fearnside 2001). At the same time, forest regrowth is a common process in smallholdings that typically implement land fallowing for soil conservation and nutrient recycling (Coomes et al. 2000). In the Brazilian Amazon, private lands are mostly concentrated in large landholdings, that have been responsible for the bulk of deforestation in the Amazon, but also hold most of the remaining forest cover (Richards and VanWey 2016). In such cases, regrowth dynamics will depend on the agricultural activity being developed, e.g., cash crops production has been found to have a negative relationship with regrowth (van Vliet et al. 2012), while ranching systems may be positively associated with forest regrowth due to land abandonment following pasture degradation in slash and burn systems (Perz and Skole 2003).

Protected areas may act as barriers to deforestation (Soares-Filho et al. 2010) and may drive forest recovery (Timm et al. 2009), although their effectiveness is co-dependent on the degree of use restrictiveness and deforestation pressure (Nolte et al. 2013); similarly, in Latin America, secured ownership and use rights in indigenous lands are known to be effective against frontier expansion pressures (Ceddia et al. 2015; Soares-Filho et al. 2010). On the other hand, insecure tenure often increases deforestation facilitating processes of land grabbing or invasions of public lands with no designated uses (i.e., undesignated) (Fearnside 2001).

Brazil offers an interesting case for investigating FCC and carbon balance outcomes under different tenure regimes. Brazil is home to the largest extent of tropical forests in the world but is also the main deforester in the tropics, because of the country's continental extent and continuously expanding agribusiness sector (Hansen et al. 2013; Margules 2004). After decades of rising deforestation rates, between 2004 and 2014 the pace of forest loss declined considerably in the Brazilian Amazon, due to fluctuations in agricultural products prices, voluntary zero-deforestation commitments by main commodity traders (e.g., soy moratorium) and public environmental policies (Assunção et al. 2015b; Moutinho et al. 2016). Notably, the Action Plan to Prevent and Control Deforestation (i.e., PPCDAm, Portuguese acronym for *Plano de Ação para a Prevenção e o Controle do Desmatamento na Amazônia*) is a multi-phase plan initiated in 2004, targeting the expansion of the protected areas network, command-and-control measures and tenure security (Arima et al. 2014; MMA 2012). However, although spatio-temporal patterns of deforestation have been extensively explored (Assunção et al. 2015b; Gollnow and Lakes 2014; Macedo et al. 2012; Richards and VanWey 2016), much less is known about net FCC within tenure categories in the Brazilian Amazon, especially prior to the 2000s (Carvalho et al. 2019a).

The increasing availability of long-term spatio-temporal information of FCC can support tropical forest carbon management, increasing our understanding of where, when and why deforestation and forest regrowth take place (Müller et al. 2016b; Schwartz et al. 2017). FCC assessments may be further combined with other datasets to provide a picture of the drivers of deforestation and forest regrowth in tropical areas, as well as accurate estimates of forest net carbon fluxes (Rudel et al. 2016; Schwartz et al. 2017; Toomey et al. 2013). One example of dataset that can be used to clarify the relationship between land tenure and FCC is the recent land registry, implemented within the Rural Environmental Cadaster database (CAR, Portuguese acronym for *Cadastro Ambiental Rural*), which filled an important knowledge gap about the spatial distribution of land tenure categories in Brazil (SICAR 2017; Sparovek et al. 2019). The CAR contains a comprehensive, georeferenced, registry of private properties' boundaries which can be further combined with already available geodata on rural settlements and public lands to provide a clearer picture of the spatial distribution of tenure regimes in Brazil.

In this paper we quantified the FCC patterns for land tenure regimes, focusing on a deforestation frontier expanding in Brazilian Southeastern Amazon, located at the influence area of the *Cuiabá-Santarém* highway (BR-163). We covered nearly three decades of FCC (1985-2012) for five categories of land tenure (designated public lands, undesignated public lands, rural settlements, private properties and lands with unknown tenure), representing the diversity of land ownership regimes present in the area (Supplementary Table II-1). The BR-163 has been a hotspot of conflict between land market developments, agricultural expansion and forest conservation, and we argue that policies, institutional changes, and tenure shifts might have had distinct impacts on FCC trajectories across tenure categories in the region. Specifically, we investigated how FCC trajectories shaped the forest carbon balance for each tenure category and reflected on their potential role as terrestrial carbon sinks and sources. We addressed two research questions:

(1) How do forest cover change patterns and temporal trajectories differ among land tenure categories in our study area?

(2) Do different land tenure categories behave as a net sinks or sources of forest cover change carbon?

# 2. Material and methods

## 2.1. Study area

The BR-163 highway channels most of the grain and soybean production of Mato Grosso (MT), the largest soybean producer in Brazil, to exporting harbors in the north of the country (Fearnside 2007). We focused on the influence area of this highway, covering a 700 kilometers track and a 100 kilometers wide buffer along the road, for which we assumed that deforestation development was influenced by the road (Fearnside 2007; Soares-Filho et al. 2006) (Figure II-1). The selected tract of the highway crosses the Brazilian states of Pará (PA) and MT - the largest deforesters in the Brazilian Amazon - in their overlapping portion with the Amazon Biome (INPE 2014b). We chose the BR-163 as study case because it has been an expanding deforestation frontier for decades, stimulated by the construction and ongoing pavement of the highway, which gave access to old-growth forests (Fearnside 2007; Müller et al. 2016a). The frontier development is concentrated after the 1980s, which allows a comprehensive reconstruction of FCC trajectories based on 30 meters (m) resolution Landsat imagery. Occupation expanded mostly from MT towards PA, which contributed to a south to north decreasing gradient in land use intensity (Soares-Filho et al. 2004)

During the last decades, several tenure changes took place, stimulated by the expansion of the frontier. Private property rights were secured as the frontier consolidated in MT and several agrarian reform rural settlements were created. Still, large tracts of public forests remained undesignated in PA, vulnerable to land grabbing (Figure II-1, Supplementary Table II-1). Conversely, in 2003, to contain deforestation outspread, a participatory plan (i.e., Sustainable Development Plan for the influence area of BR-163) led to the creation of a sustainable use forest district (Brasil 2006), with mixed results (Schönenberg et al. 2015).

In this context, in the mid-2000s, already under the PPCDAm, the protected areas network surrounding the highway was expanded, with the creation of national forests for sustainable timber extraction (e.g., Flona Jamanxim, Flona Altamira; Flona is the Portuguese acronym for National Forest), extractivism reserves and strict use conservation units (Figure II-1; Supplementary Table II-1).

## 2.2. Land tenure categories

We distinguished five categories of land tenure, implicating in distinct land use management systems and use restrictiveness levels. These were: (1) designated public forests; (2)

undesignated public lands; (3) rural settlements; (4) private properties and (5) lands with unknown tenure (Figure II-1; Supplementary Table II-1).

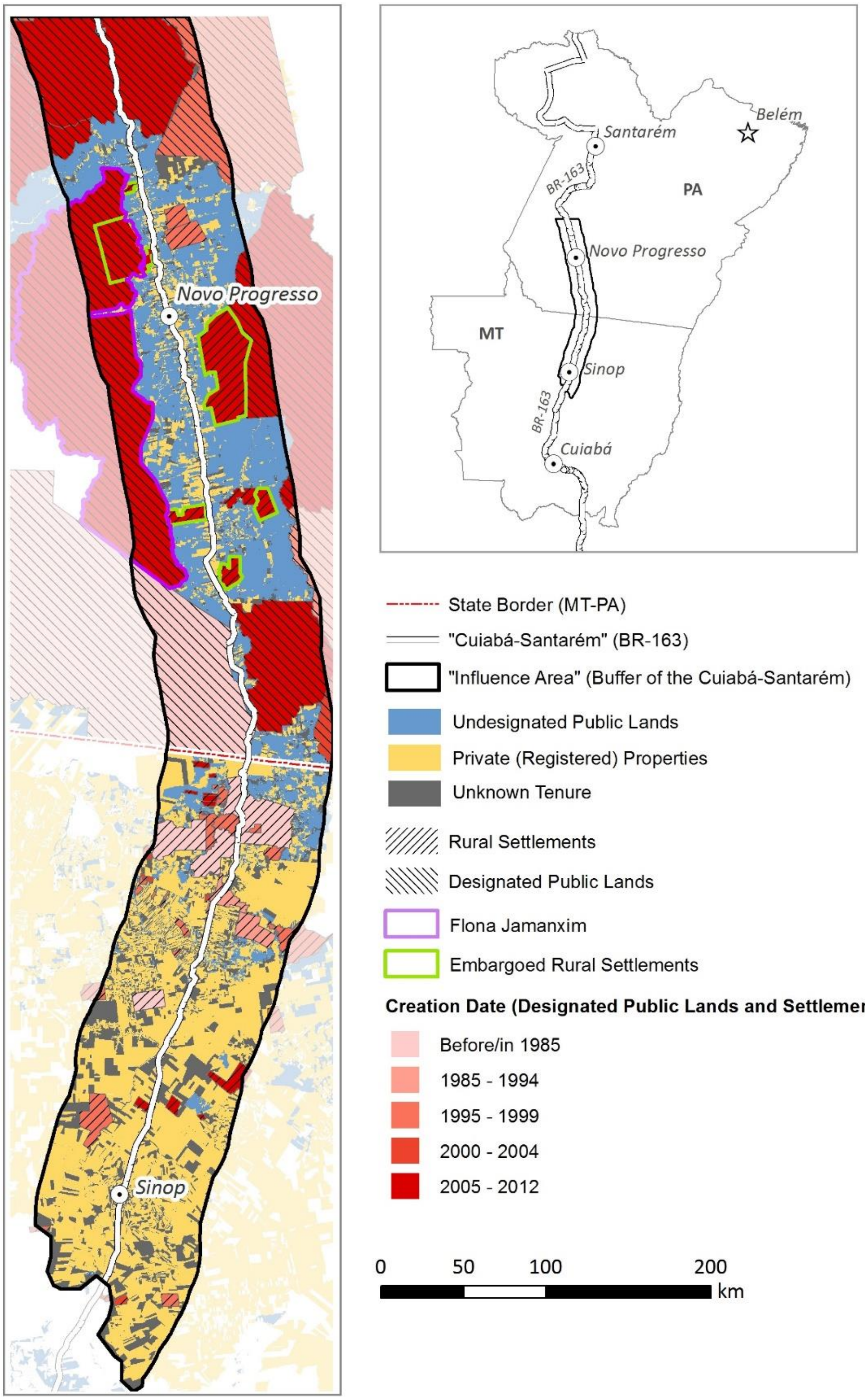



**Figure II-1. Study area.** Influence area of the BR-163, between Sinop (MT) and Novo Progresso (PA) and land tenure categories in 2012. Color gradient indicates the creation year of conservation units, indigenous lands and military areas included in the "designated public forests" category, and of "rural settlements" category.

Designated public lands include state-owned lands which have received sustainable use, strict nature protection or military use designations. In strict protection categories, land use is not allowed and access is controlled by the state; in conservation units for sustainable use few economic activities are permitted (e.g., timber and non-timber forest products extraction) with use or withdrawal rights granted to third parties (e.g., timber extraction companies, traditional communities) via concessions and oversaw by the state. We included indigenous lands in this category, defined as community property regimes governed by native populations and from which non-indigenous groups are excluded from access, use, and resources withdrawal rights. We included indigenous lands in this category because land uses typically have low environmental impacts, plus land transactions are not allowed; however, invasions for mining, cattle ranching, or cash crops production may occur. Between 1985 and 2012, the area covered by designated public lands was restricted to the state of PA and increased four times, reaching 24,184 $km^2$ by 2012.

Undesignated lands are state-owned lands for which no land use was designated. In principle, all other social actors are excluded from access, use, withdrawal and alienation rights, but, in context specific cases, traditional populations and longstanding settlers may secure tenure via legal avenues. However, with loose governance, undesignated lands and associated resources (e.g., commercial timber) are constantly under dispute by individuals or groups seeking to exclude each other from land use privileges. Therefore, undesignated lands are the main target of land grabbers and illegal deforestation. Typically, irregular claimant users will not observe land use limits established by law and are less responsive to policies that require formal ownership, leading to deforestation and forest degradation. In the last decades large tracts of public lands have been assigned to specific tenure regimes, either settlements, conservation units or private ownerships. In our study area, we tracked a decrease in undesignated lands of 52.9% in MT and 63.1% in PA providing space for rural settlements and strict use areas (Supplementary Table II-1).

Rural settlements are groups of rural landholdings created as agrarian reform projects on public or expropriated lands to promote smallholder livelihoods. There are distinct settlement designs, implemented through private property regimes in which settlers are entitled all but alienation rights over their parcels (until payment for lands is completed) or collective property regimes in which settlers are granted community titles. Settlers have the duty to use their lands and manage common-pool resources (i.e., forest reserves named as "Legal Reserves" – LRs - by the Brazilian Forest Code - BFC), observing land use restrictions established by the BFC. In forestlands of the Amazon biome the BFC limits use quotas to a maximum of 20% of the lot, but, in case of forest shortage, the required LR equals to forested area by 2008 (Brasil 2012). Parcel size is defined based on the amount of land necessary for family subsistence (~100 hectares in the study region), and typical land uses are cattle ranching, extractivism, shifting agriculture or, to a smaller extent, cash crops production. Between 1985 and 2012, rural settlement area expanded in our study area by 94.6% in MT (2,221.0 $km^2$ of expansion); in PA, by 1985 no settlements had been implemented, but by 2012, 2,780 $km^2$ had been assigned to agrarian reform settlements in our study area.

In private properties, owners hold all rights (access, use, withdrawal and alienation) and the duty to restrict land use to the limits established by the BFC. Like in rural settlements, use quotas equal to a maximum of 20% of the lot area, with the remainder of forest stocks being kept as LRs. In smallholdings, the required LR equals to forested area by 2008, but in larger properties, in case of LR forest shortage, restoration requirements will vary as described by the BFC. Private properties develop several rural activities, from cash crops production in intensive double-cropping systems, especially concentrated in the southern plateaus of MT, suitable for mechanization, to cattle ranching, produced in semi-confined to extensive systems. Finally, we defined as "unknown tenure" all lands for which we had no information about land tenure, which could be either private and not registered in the CAR or public and not included in any datasets.

We did not account for conflicts between tenure designations (i.e., overlaps between tenure categories in used datasets) although they exist (Sparovek et al. 2019); instead, we used a hierarchic approach (Supplementary Table II-1, first column), comparing the jurisdictional prerogatives of each tenure system to decide which should take priority in the case of overlap (Freitas et al. 2016) (e.g., designated public lands prevails over rural settlements in case of overlap). The data sources of the spatial layers combined to create the tenure category map are listed in Supplementary Table II-1. Land tenure changed across the period, as public lands were designated to conservation, to agrarian reform settlements and private ownership. Since we could not track such changes for all tenure categories[10], we opted to use current land tenure information and kept tenure categories static while tracking FCC and associated carbon fluxes. However, we contrasted the FCC trends found with known tenure changes (Supplementary Table II-1) in the discussion section to support the discussion of our results.

## 2.3. Forest cover change and carbon modeling

[10] Specifically, we could not track the assignment of private tenure to individual properties established over non-designated lands.

We quantified annual FCC, i.e., deforestation, forest regrowth and forest re-clearance, for each tenure category, separately for the states of MT and PA. We used annual forest cover change maps between 1985 and 2012 obtained from Müller et al. (2016a) and Müller et al. (2016b) with a 30 meters pixel size, produced using Landsat TM/ETM imagery. To avoid overestimating FCC, we chose a minimal mapping unit of forest loss and regrowth of one hectare and used a clump algorithm to remove smaller events. FCC dynamics were only assessed for forestlands. Therefore, we masked out non-forest areas (e.g., savanna or outcrops) using a forest mask provided by the Brazilian Amazon deforestation monitoring program (PRODES) (INPE 2014b). For more details on the FCC dataset please refer to the original references of Müller et al. (2016a) and (Müller et al. 2016b); for details on the FCC dataset post-processing see Hissa et al. (2016) (Appendix Chapter).

We calculated gross and net annual forest carbon changes applying the FCC maps to a pixel-level carbon-bookkeeping model (Aguiar et al. 2012; Houghton et al. 2000). This model calculates the amount of carbon stored in woody biomass that is released by deforestation. We worked under the assumption that a share of the biomass is burned, and its carbon is released immediately. The remainder is gradually released in subsequent years, depending on the carbon decay time-lag associated to four different pools (i.e., elemental carbon, wood products, residual slash and roots). Carbon uptake of forest regrowth was modeled linearly at a 1.2% a year recovery rate (Lennox et al. 2018). We assume that carbon stored in regrowing forests is immediately released to the atmosphere following re-clearance due to the common use of fire. Please refer to Aguiar et al. (2012); Hissa et al. (2016) (Appendix Chapter) and the supplementary material (Supplementary Table II-2) for details on the carbon bookkeeping model parameterization.

# 3. Results

Tenure categories were distributed heterogeneously in our study area (Figure II-1, Supplementary Table II-1). Designated public forests and "private properties" were the most representative categories, covering 29.7% (24,227 $km^2$) and 29.6% (24,174 $km^2$) of the area, respectively, followed by undesignated public lands (18.3%; 14,938 $km^2$) and rural settlements (9.0%; 7,346 $km^2$). For 13.5% of the area (10,986 $km^2$) there was no tenure information available (Supplementary Table II-1). MT and PA also have markedly different tenure systems: while in MT "private properties were the main tenure category, in PA, the designated public forests class prevailed, followed by the undesignated public lands category (Supplementary Table II-1).

## 3.1. Old growth forest losses

We found differences in FCC patterns for distinct land tenure categories (Supplementary Table II-1, Figure II-2a-d), which were stronger for old-growth deforestation. Between 1985 and 2012, nearly half of deforestation of old-growth forest occurred on private properties (47.6%; 12,546 $km^2$), of which 77.4% (9,720 $km^2$) took place in MT. 21.0% (5,535 $km^2$) of old-growth forest losses was observed on lands with unknown tenure, most of which took place in MT (71.8%; 3,974 $km^2$). Fifteen percent of forest losses occurred in rural settlements (3,987 $km^2$), and were largely concentrated in MT (75.6%; 3,015 $km^2$). Pará concentrated losses in undesignated public lands (79.1%; 2,252 $km^2$) and designated public forests (99.9%; 1,435 $km^2$). Both the extent and change in forest cover were found to be negligible in the designated public forests category in MT.

Strikingly, by 2012, rural settlements in MT had lost 84.9% of the old-growth forest cover, followed by private properties in PA (81.8%) and MT (63.3%) and lands with unknown tenure in PA (71.3%) and MT (65.7%) (Supplementary Table II-1, Figure II-2a-b). Despite the high absolute losses, designated public forests and undesignated public lands in PA still held a large share of old-growth forests in 2012, due the large extent of old-growth forests in these tenure categories (Supplementary Table II-3, Figure II-2b). In private properties and lands with unknown tenure of MT, deforestation oscillated but was concentrated in the period between the mid-1990s and mid-2000s, decreasing after 2004. On the other hand, in rural settlements and undesignated public lands we observed a delayed and diffuse downward trend in deforestation rates by the end of the period (Supplementary Figure II-1). Deforestation was concentrated in the second half of the studied period for all categories in PA (Supplementary Figure II-2) where deforestation rates continued to rise in rural settlements and undesignated public lands until 2012.

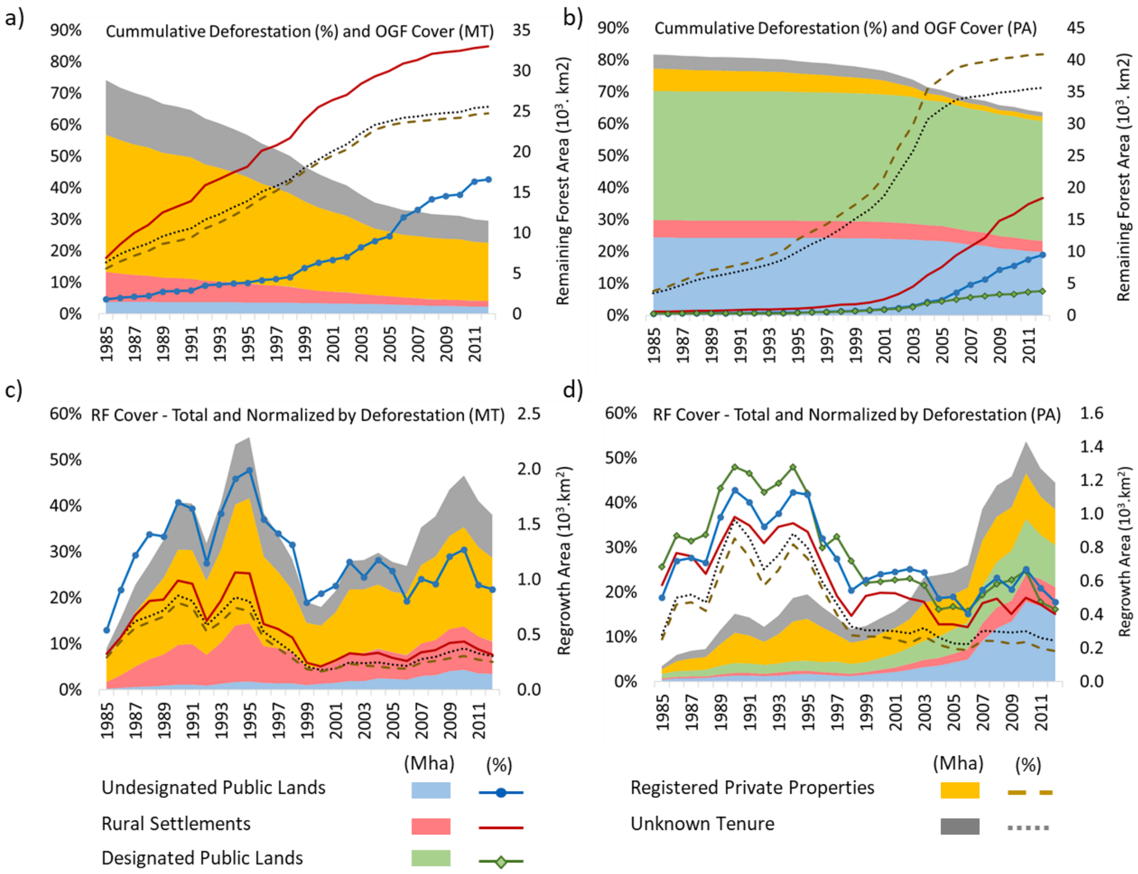



**Figure II-2. Forest cover change (FCC) outcomes for tenure categories.** Lines in Figures a-b show accumulated old-growth forests clearing relative to the original forest cover (%) and solid areas show remaining forest cover (km²). In Figures c-d, lines show the percentage of deforested area covered by forest regrowth (%) and solid areas show regrowing forest cover area (RF Cover, km²).

## 3.2. Forest regrowth and net forest cover change

By 2012, regrowing forests represented 6% of the total forest cover (2,768 km$^2$), 12.1% in MT and 3.6% in PA (Supplementary Table II-3). In categories where old-growth forests were severely depleted, regrowing forests already represented a large share of total forest area (e.g., 30.1% in rural settlements) (Supplementary Table II-3, Supplementary Figure II-1a-h and Supplementary Figure II-2a-j). In MT, private properties concentrated 47.9% of the regrowing forests (759 km$^2$) (Figure II-1c). For most tenure categories in this state, forest regrowth area peaked around the mid-1990s, except in undesignated public lands, where regrowth area expanded continuously (Figure II-2c). In PA, regrowing forests expanded intensively in the second half of the study period. In 2012, undesignated public lands contained 34.7% of total regrowth in PA (411 km$^2$), with the remainder relatively well distributed among the other tenure categories (Supplementary Table II-3, Figure II-2d).

Regrowth prevalence in deforested areas was more prominent in PA (12.2%) than MT (7.1%). We observed a marked decrease in the proportion of deforested areas covered by regrowth from the mid-1990s onwards for all categories and in both states (Figure II-2c-d). Undesignated and designated public forests showed a larger proportion of regrowth in deforested areas than other tenure categories, ranging between 40-50% around the mid-1990s and stationing around 20% towards the end of the period. In MT private properties, unknown tenure and rural settlements showed less regrowth prevalence, decreasing from a 25-15% range to less than 7-6% by 2012; In PA regrowth prevalence ranged from 35-20% for these classes in the mid-1990s to 15-7% by 2012.

## 3.3. Forest cover change carbon balance

Even though we detected years of very subtle net forest cover increase (Supplementary Figure II-1a-h; Supplementary Figure II-2a-j) all tenure categories in both states behaved as carbon sources throughout the study period (Figure II-3a-b). By 2012, a total of 1,541.8 $TgCO_2e$ had been lost due to old-growth forest losses and 31.7 $TgCO_2$ to regrowing forests removal, totaling 1,573.0 $TgCO_2e$. Only 40.3 $TgCO_2e$ (2.6%) of total losses were offset by forest regrowth, yielding a net FCC balance of 1,532.7 $TgCO_2e$. In sum, private properties were the total largest net emitters in MT (567.0 $TgCO_2e$) and PA (177.0 $TgCO_2e$), followed by unknown tenure in MT (232.3 $TgCO_2$) and undesignated public lands in PA (120.1 $TgCO_2e$).

We compared the mean annual net emission values for the periods between 1996-2005 (i.e., the time period used as baseline in Brazil's NDC and the National Climate Change Policy) and 2006-2012. Private properties and lands with unknown tenure showed the most intense decrease in emissions. In MT, we found a 57.8% and 55.0% reduction in emissions from private properties and lands with unknown tenure, respectively, and a more moderate decrease in PA, of 28.4% and 25.7% for the same categories; rural settlements in MT also showed a reduction of 48.0%. On the other hand, all other categories underwent high increase in mean emissions between 2006-2012, compared to 1996-2005, that reached 682.7% and 117.6% for undesignated lands in PA and MT, respectively, 319.1% for settlements in PA and 163.3% for the designated public forests category in PA. In the year of 2012, categories showing increase in emissions in PA were already accounting for nearly 80% of total FCC net emissions (Figure II-3d).

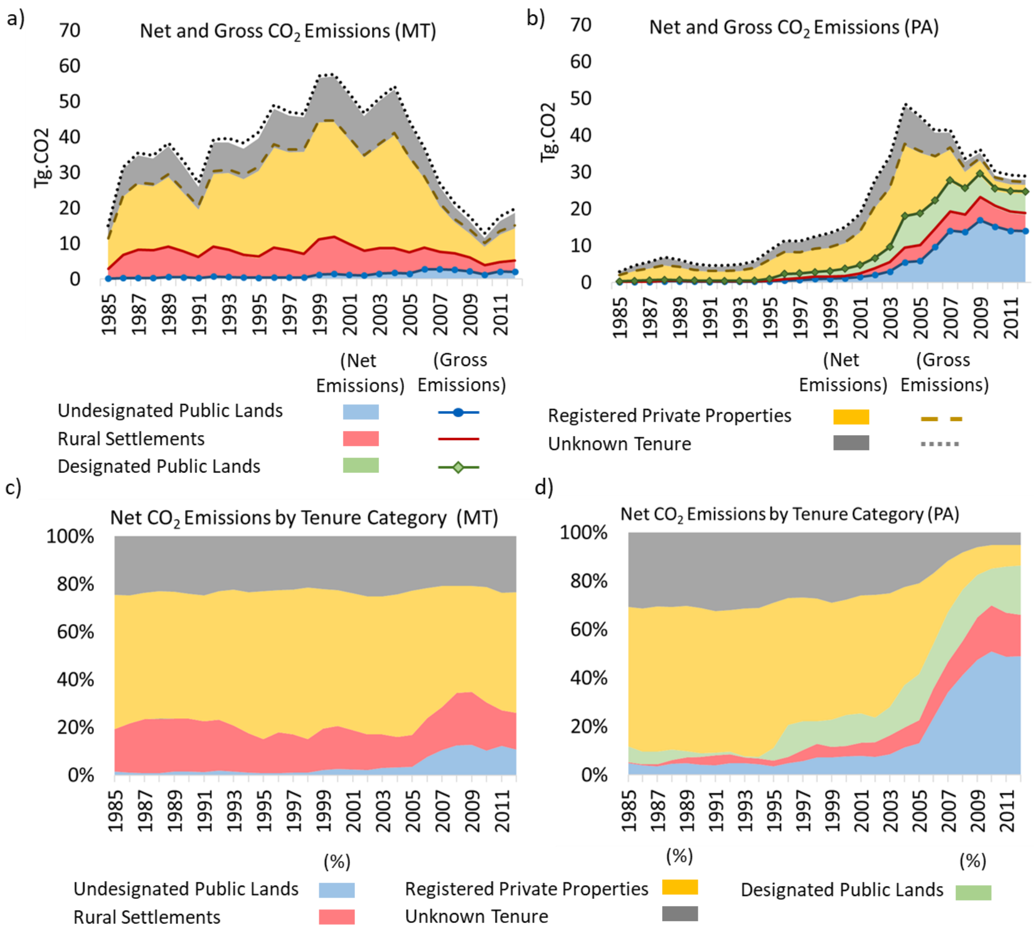



**Figure II-3. Net and Gross $CO_2$e emissions.** Lines in Figure a-b show annual gross emissions (Tg.$CO_2$e) from old-growth forests deforestation and regrowing forests re-clearance and solid areas show annual net emissions from forest cover change (FCC). Figures c-d show the relative (%) share of each land tenure category, annually, on net emissions from FCC.

# 4. Discussion

## 4.1. Old-growth forest losses

Our results showed high absolute and relative losses of old-growth forests between 1985 and 2012 and indicated distinct FCC outcomes for tenure systems across time. Old-growth forests stocks located in private properties were severely depleted in both states, confirming findings by previous studies focusing on different areas in Latin America, which found private properties as both the largest deforesters and holders of remaining forests (Fearnside 2005; Michalski et al. 2010; Paneque-Gálvez et al. 2013; Richards and VanWey 2016). On the other hand, most of the deforestation reduction detected from the mid-2000s onwards took place in private properties, both in MT and PA (Figure II-2a-b; Supplementary Table II-3).

Conversely, in PA, deforestation rates in public forests, designated or not, and rural settlements increased during the 2000s and remained high until 2012.

These results suggest that private landholders with registered land holdings were responsive to last decade's policy efforts to reduce deforestation (Arima et al. 2014). The CAR is the basis for the BFC enforcement in private lands. It allows the monitoring of land use and cover change within properties to verify law compliance and access to credit and farm's produce transactions are increasingly bound to the registry to support the BFC enforcement (Klingler et al. 2017). Additionally, if landholders adhere to the CAR registry before the deadline established by decree[11], they may be granted access to laxed conditions for compliance with the Brazilian Forest Code (BFC), e.g., reducing liabilities of non-compliers (Brasil 2012; Sparovek et al. 2012). Therefore, one plausible explanation for the deforestation reduction in those areas is that the bundle of public and private policies to reduce deforestation raised deforestation opportunity costs amongst private landholders that depend on a legality (or compliance) status to obtain credit or sell their production (Assunção et al. 2016; Börner et al. 2014; Nepstad et al. 2014) – and that the registry may be a good indicator of such actors. Although the CAR is a prerequisite to operate in conformity, joining the registry depends on the initiative of each landholder, which may indicate a predisposition to abstain from illegal deforestation. Therefore, property registration has been linked to deforestation reduction in PA and MT (Alix-Garcia et al. 2017).

For both states, lands with unknown tenure showed similar FCC behavior as private properties, i.e., following the trend in decreasing deforestation and carbon emission rates (Figure II-2a-b, Figure II-3a-b, Supplementary Table II-2). As Freitas et al. (2017b), we interpret this as an indication that these lands are likely privately owned (or are to be claimed as such) although still unregistered, for large tracts of land with unknown tenure, and that the benefits of anti-deforestation policies were obviously not restricted to registered private properties. However, by the time we collected the property boundaries data (December 2016) the deadline for registration to the CAR system was still open, and such landholders could have registered later, making it hard speculate on the motivations of social actors based solely on the property registration adherence.

Three (non-exclusive) causes could explain why rural settlements deforestation remained high in PA: the first is related to settlements age, the second to the deforestation alert system

[11] The original deadline for registering the rural properties in the CAR system was set to 31st of May 2015, However, the deadline has been pushed forward several times and is currently set to end on 31st of December 2020.

and the third to settlers' land use behavior in the context of recent colonization policies. First, the expansion of rural settlements in southwestern PA is recent (Figure II-1, Supplementary Table II-1) and more forest stocks were available for clearing at the time of their establishment during the mid-2000s than in settlements of MT (where accumulated deforestation crossed the 75% mark by 2004). Second, in the Amazon, a satellite-based alert system provides inputs for field-based law enforcement campaigns, a strategy that proved to be more effective in inhibiting medium to large sized forest clearings than small-scale deforestation (< 20 hectares) (Borner et al. 2015) typical of smallholdings. Therefore, it is possible that much of the deforestation in settlements was overlooked by alert systems (Assunção et al. 2015a). Third, very often, the establishment of settlements is not followed by investments in infrastructure, driving settlers to sell the commercial timber in their lots to illegal loggers, which in return open roads (information based on interviews conducted by the first author during field work in Novo Progresso, Pará in July 2015). This, combined with the fact that settlers are motivated to quickly convert forests into productive lands to secure tenure, leads to widespread deforestation (Fearnside 2001; Peres and Schneider 2012). Many times, landless people allocated to settlements have no previous experience in subsistence agriculture and are left without technical support; they frequently turn to cattle ranching implemented via slash and burn systems that quickly degrade the land (Alencar et al. 2016; Peres and Schneider 2012). Consequently, they often abandon (i.e., leaving the land up for grabbing), rent or illegally sell their lots to neighboring ranchers (information based on interviews conducted by the first author during field work in Novo Progresso, Pará in July 2015). In fact, several agrarian reform settlements created in PA in the mid-2000s, six overlapping our study area (Figure II-1), were embargoed by the Federal Prosecution Office of Pará that questioned the environmental and technical feasibility of the projects. Settlers in embargoed projects have no access to credit, financing or technical support and are easy targets for landgrabbers and loggers, which could have fueled deforestation (Alencar et al. 2016) – however, no study has formally tested this hypothesis so far.

Old-growth forest losses continued to rise in public lands, both undesignated and designated (Supplementary Figure II-1a-b, Supplementary Figure II-2a-b, Supplementary Figure II-2e-f). Historically, the illegal occupation and deforestation of undesignated public forests has been a typical mechanism of informal colonization in the Amazon (Araujo et al. 2009; Azevedo-Ramos and Moutinho 2018). Landgrabbers deforest with the intention of selling illicitly occupied public lands and will often dispute them with titleless traditional populations, claiming land rights either using violence or distorting legal avenues to do so

(Aldrich et al. 2012). Along the BR-163, one additional hypothesis might explain the surge in deforestation in undesignated forests during the last decade, especially in PA: around the mid-2000s, the pressure for compliance in lawful properties decreased the availability of land available for agricultural expansion (mostly pastures) which spurred deforestation in undesignated lands.

The continuous increase of deforestation in the designated public forests category may be largely explained by the FCC dynamics in the Flona Jamanxim, a conservation unit for sustainable forest management created in 2006 encompassing 130,000 km$^2$ (of which, 12,345 km$^2$ overlap our study area) easily accessible via the BR-163 (Figure II-1). The creation of this conservation unit was always a matter of heated debate by the local population that accuses the government of imposing the limits of the area without proper participatory consultation, arguing that more rightful occupations by titleless landholders were previously present than acknowledged by the federal government (Campbell 2015, Klingler et al., 2017, and information based on interviews conducted by the first author during field work in Novo Progresso, Pará in July 2015). A proposal to downgrade and downsize the limits of the Flona Jamanxim to less than half of its original extent was vetoed in 2016 by the former President Michel Temer after mounting pressure by the media and civil society (Brasil 2016a). Such scenario of tenure insecurity could have motivated inhabitants to deforest to proof occupation and have attracted squatters aiming to take advantage of the instability to claim lands. As a result, the Flona Jamanxim is listed as one of the conservation units most affected by deforestation in Brazil and in the world (Araújo et al. 2017; Collins and Mitchard 2017).

### 4.2. Forest regrowth, net forest cover change and carbon balance

Overall, regrowing forests cover expanded fivefold since 1985. Still, FCC showed a small potential to offset carbon losses driven by deforestation, across states and tenure categories (Figure II-3a-b; Supplementary Table II-3). Net deforestation rates were either higher or similar to regrowth rates and recovering forests showed a low average permanence time (~ 5 years) (Supplementary Figure II-3). Even in the few years in which forest expansion surpassed net forest losses all tenure categories still behaved as carbon sources because legacy emissions from deforestation in previous years exceeded gains from carbon uptake due to forest regrowth. Our study area showed a behavior typical of an expanding deforestation frontier in which forest expansion and losses are coupled, occurring sequentially and simultaneously, driven by the dynamics of the productive systems (Arvor et al. 2016; Sloan

2008). There were no signs that a forest transition is underway in any of the tenure categories, and even less driven by sustainable transition pathways (Aguiar et al. 2016).

Public lands and rural settlements showed more relative forest regrowth than private properties and lands with unknown tenure, a pattern identified in both states (Figure II-2a-b). Despite the absolute increase in regrowing forests area this land cover progressively occupied a smaller share of deforested lands, indicating an ongoing process of land use intensification (Carvalho et al. 2019a; Müller et al. 2016b) (Figure II-2c-d). Tenure regimes and shifts may have contributed to reduce relative regrowth and can partially explain differences between tenure categories. For example, the creation of new settlements between 1994 and 1997 coincides with a downward trend in relative regrowth area (Supplementary Table II-1, Figure II-2c-d). This shift in tenure likely changed local land use behavior towards agricultural or pastoral uses in which regrowth occupies less area and persists for less time in the landscape opposite to the cryptic and abandonment-prone occupation characteristic of insecure tenure lands (Aguiar et al. 2016; Aldrich et al. 2012).

The decrease in relative regrowth was most accentuated in private properties, which also showed less regrowth prevalence in deforested areas among tenure categories. In 1996, a revision of the BFC increased the protection cap in private properties from 50% to 80% (Stickler et al. 2013); landholders, confronted with this additional land restriction and forthcoming anti-deforestation policies, might have chosen to expand agricultural production over fallowing or abandoned lands, instead of old-growth forests, reducing the area under regrowth (Carvalho et al. 2019a).

However, more importantly, land use dynamics in private properties of the Brazilian Amazon are increasingly coupled with international commodity markets and, hence, cannot exclusively be explained by local drivers (Nepstad et al. 2006; Pacheco and Poccard-Chapuis 2012). For example, although human labor is reduced in industrial-scale agriculture, as developed in the surroundings of Sinop/ MT (Figure II-1), these areas still support little (relative) forest regrowth (Figure II-1c-d). This development is contrary to the rationale that has linked rural population decline to a resurgence in forest cover (Mather 1992; Mather and Needle 1998), which, may be the dominant case in temperate regions, but not in the tropics (Lambin and Meyfroidt 2010; Rudel et al. 2005; Rudel et al. 2016; Sloan 2007; Sloan 2008). Instead, by the late 1990s, FCC in MT was decoupled from population dynamics (Perz et al. 2005) and largely driven by the demand for cash crops and cattle products, which fueled deforestation (Nepstad et al. 2006), but little forest regrowth.

Additionally, in private properties, high forest fragmentation is increasing the distance to propagules, and prolonged and intensive land management (e.g., use of fire, less frequent fallowing, use of fertilizers and mechanized soil preparation) is destroying soil seed banks (Vieira and Proctor 2007). Such processes reduce regrowth by impairing the ability of the land to undertake forest regeneration naturally if agricultural activity ceases (Sloan 2016; Wandelli and Fearnside 2015).

On the other hand, relative regrowth was consistently higher in undesignated and designated public lands. Insecure tenure and restricted access to credits by land users within this category may have made them more reliant on frequent fallowing for soil nutrient recovery and less likely to invest resources in prevent shrub and woody encroachment over pastures. If deforestation takes place for land speculation (as it is common in public lands) and not for agricultural production, abandonment and regrowth are also common (Pacheco 2012).

There was a synchronous decrease in regrowth relative to the deforested area between 1995 and 2000 across all tenure categories (Figure II-2c-d). Müller et al. (2016b) argued macroeconomic and environmental processes taking place at larger scales could have produced this outcome. The implementation of a new Brazilian currency (the Real) in 1994 initially boomed deforestation, but Fearnside (2002) argued that the decrease in inflation from 1995 onwards reduced the role of land acquisition as a protection against currency depreciation, reducing deforestation motivated by speculation. It is possible that the reduced inflow of land into land markets drove the use of previously opened lands, consequently, reducing regrowth. This could help explain why poorly governed lands (e.g., undesignated public lands) also showed a decrease in relative regrowth: although these areas are not as responsive to the BFC rules, they are very responsive to changes in profitability linked to land speculation. The intense El-Niño droughts between 1997/1998, caused water deficit and likely favored fires, which could have caused grass and shrublands encroachment and arrested woody vegetation recovery in the following years.

Average age of regrowth was low and consistent with numbers found by studies that used long time series of FCC data (Supplementary Figure II-3) (Almeida 2008; Schwartz et al. 2017) and has shown little divergence among tenure classes. The younger age of regrowing forests in undesignated forests and rural settlements in PA is likely caused by the newer occupation history and is characteristic of an active deforestation frontier, bringing down the average age of forest regrowth.

### 4.3. Implications for policy

The Brazilian strategy for REDD+ (Reducing Emissions from Deforestation and Forest Degradation) is largely based on demonstrated achievements in reducing deforestation, which are threatened by insecure tenure (Correa et al. 2019). By 2012, in our study area over 10,000 km$^2$ of old-growth public forests lacked designation, a recurrent pattern in the Amazon (Moutinho et al. 2016). The mutual occurrence of high deforestation and forest regrowth in undesignated public lands indicate a high land and deforestation sparing potential. If ownership is rightful in undesignated lands, it is important to secure tenure and promptly enforce the BFC (Freitas et al. 2017a). Where it is not, the allocation of undesignated lands for protection or sustainable use has been raised as a pivotal strategy to reduce deforestation and achieve climate change mitigation commitments (Azevedo-Ramos and Moutinho 2018; Freitas et al. 2017a). However, this process should be based on studies and participatory consultation involving federal or state governments (or both, where pertinent) and relevant actors across scales, to minimize conflicting claims, as the example of the Flona Jamanxim demonstrated.

Our results suggest a lack or inefficacy of policy instruments to motivate (high quality) forest regrowth beyond what is driven by business as usual land use dynamics. For example, recovery of (illegally) deforested lands did not increase, following the designation of public lands to conservation and sustainable uses during the 2000s. Similarly, by 2012, relative regrowth decreased in private properties and rural settlements and was kept at a young age (Supplementary Figure II-3), adding a very small contribution to offset shortages in the forest cap required by the BFC at the time (80% of the landholding). In fact, a recent attempt to regulate forest regrowth management in Pará (State Law – IN-02 of 2014) backfired as private landholders reduced regrowth permanence time to avoid the need of licenses to re-clear land (information based on interviews conducted by the first author during field work in Novo Progresso, Pará in July 2015). Therefore, to achieve forest restoration targets in the Amazon it is necessary to address the many technical and institutional challenges of governing forest regrowth (Vieira et al. 2014), mobilizing social actors and improving and creating new instruments, tailored for tenure systems (Bustamante et al. 2019). In this context, FCC monitoring may offer verifiable indicators that can be assessed remotely and validated on the ground to help address information gaps about the quality of forest regrowth.

### 4.4. Shortcomings

Tenure security is the entry point for several other policies that may have positive or negative impacts on net FCC. Therefore, it is not simple to isolate the impact of tenure on FCC and it was not our objective to estimate the magnitude of the influence of tenure systems as drivers of FCC; we rather identified different patterns of FCC across time and gathered information on policies related to tenure that may have influenced and help elucidate our findings.

Land tenure systems are dynamic in frontier areas, which has obvious implications for our assessment. Unfortunately, detailed information on ownership regime changes across our study area was not available for all tenure classes in our study area. By using a static tenure map, we downplayed the historical relevance of undesignated public forests in FCC, since most tenure categories originated from undesignated public forests at some point during our period of analysis. We addressed this source of uncertainty in this discussion session, highlighting the known shifts in forest tenure (Supplementary Table II-1) and likely implications for the FCC patterns we found.

## 5. Conclusions

The influence area of the BR-163 is a dynamic region that exhibited heterogeneous FCC outcomes for tenure regimes across space. We identified a marked reduction in deforestation and carbon emissions in private properties coinciding with the efforts to reduce deforestation in the mid-2000s. The reduction was more intense in MT, an important outcome for conservation, given that most forestlands in this state are privately owned. On the other hand, forest losses in undesignated public lands increased during the 2000s, suggesting that lands with insecure tenure were not responsive to anti-deforestation policies, or worse, that deforestation leaked from areas where policies were effective, a hypothesis that future research can evaluate. Hence, it is worrisome that large tracts of forests remain undesignated in PA and to a lesser extent, in MT. Our results endorse recommendations about the need to regularize tenure in undesignated public lands for a steady decrease in deforestation and carbon emissions in the Brazilian Amazon. Shifts in tenure regimes influenced trajectories of FCC and associated outcomes. In PA, the establishment of rural settlements after the 2000s was followed by intense deforestation. In the same state, we speculate that the creation of a sustainable use conservation unit in 2006 failed to contain forest losses and could have led to an increase in deforestation in the designated public lands category due to tenure conflicts.

Therefore, it is important to acknowledge that securing tenure is not enough to control deforestation if land designation is not accompanied by effective law enforcement and assistance (e.g., technical consultation to settlers).

Carbon uptake via forest regrowth was consistently low across tenure categories and showed little potential to offset carbon losses from net deforestation. Our findings suggest that forest regrowth taking place along the BR-163 is likely a product of the agricultural land use systems and is not related to a resurge in forest cover following frontier consolidation. Just the opposite, regrowth has a short duration on the landscape and is progressively occupying less (relative) space. The decrease in regrowth presence in deforested lands could be positively interpreted as a reduction in land vacancy that, to some extent, spared old-growth forests from deforestation. However, taking the BR-163 influence area as an example, we also conclude that it will be a great challenge for future policies to incentivize forest recovery in the Amazon, in all tenure regimes, to achieve forest restoration targets and LRs reestablishment.

We conclude that terrestrial carbon management for the purpose of climate change mitigation should account for land tenure effects on processes of deforestation and forest regrowth. We stand by the importance of using long-term spatial time-series of FCC trajectories to provide insights about the impacts of past policies on forest losses and recovery. Because our study was temporally comprehensive (1985-2012) and based on annual land cover change estimates we were able to associate changes in FCC trajectories and carbon emissions with specific policies, tenure shifts and local to macro-scale changes more precisely. Such approach can aid policymaking, tailored for specific social actors and tenure regimes, increasing success rates and anticipating failures.

# 6. Supplementary material

**Supplementary Table II-1. Land tenure categories.**

| Hierarchy + Tenure System | Coverage (km²)* | Description | Land Uses and Processes | Data Sources | Known Tenure Shifts |
|---|---|---|---|---|---|
| **(#1)** Designated Public lands | (MT) 42.6 (0.1%)<br><br>(PA) 24,184.8 (52.1%) | *Conservation Units/ Military Areas* include state property regimes with designated uses for nature conservation, protection or military use.<br>*Indigenous lands* include community property regimes in which non-indigenous groups are excluded from use rights. | Strict protection, sustainable uses (timber and non-timber forest products extraction via concessions), subsistence shifting agriculture in indigenous lands, typically extensive systems.<br>Invasions for mining, extensive cattle ranching, and cropping may occur. | Includes Conservation units (except for APA category), Military areas and Indigenous Lands established by 2012.<br>*Sources*: Ministry of Environment (MMA) http://mapas.mma.gov.br/ Indigenous People National Foundation (FUNAI) http://www.funai.gov.br | Strict use areas remained static in MT.<br>In PA, strict use areas increased fivefold between 1985 and 2012, areas were most likely sourced from undesignated public lands<br>**1985** 4,789.7 **2006** 22,639.2<br>**1998** 6,750.4 **2008** 24,184.8<br>**2002** 7,077.6 **2012** 24,184.8<br>**2005** 10,293.8 |
| **(#2)** Rural Settlements | (MT) 4,565.8 (13.0%)<br><br>(PA) 2,780.0 (6.0%) | Private property regimes or community property regimes in which public or expropriated lands are designated to promote agrarian reform and smallholder livelihoods. Settlers are entitled all but alienation rights over their parcels and have the duty to use their lands and manage common-pool resources (i.e., Legal reserves), observing land use limits established by the Brazilian Forest Code (BFC). Depending on the design of the settlement, use quotas equal to a maximum of 20% of the lot. If not, in case of forest shortage, required Legal Reserve equals to forested area by 2008. | [Permitted, when registered and committed to compliance to the BFC]<br>Cattle ranching, subsistence shifting agriculture; slash and burn, typically, extensive systems. Sustainable uses (timber and non-timber forest products extraction).<br>Landowners/holders often don't comply to the BFC.<br>Land abandonment, irregular alienation and contestations may occur | Includes Agrarian Reform Rural Settlements established by 2012.<br>*Source*: National Institute for Colonization and Agrarian Reform www.incra.gov.br (INCRA) | In MT, most expansion was prior to the 2000s, likely sourced from undesignated public lands<br>**1985** 2,344.8 **2007** 4,304.0<br>**1995** 3,500.4 **2012** 4,565.8<br>**1998** 4,020.4<br>In PA, most expansion occurred during the 2000s, likely sourced from undesignated public lands<br>**1985** 0.0 **2005** 1,531.7<br>**1996** 161.3 **2012** 2,780.0<br>**1997** 444.4 |

| | | | | | |
|---|---|---|---|---|---|
| **(#3)**<br>Registered Properties | (MT)<br>20,294.1<br>(57.6%)<br><br>(PA)<br>3,879.4<br>(8.4%) | Private property regimes in which individual owners hold all rights (access, use, withdrawal and alienation) and the duty to restrict land use to the limits established by the BFC. Use quotas equal to a maximum of 20% of the lot. Required Legal Reserve equals to forested area by 2008 if the property is a smallholding. If not, in case of forest shortage, restoration requirements will vary as described by the BFC. | [Permitted, when committed to compliance to the BFC]<br>Cattle ranching, agriculture (large and small scale), Sustainable uses (timber and non-timber forest products extraction); low-to-high intensity systems.<br>Landowners/holders often don't comply to the BFC. | Includes private properties submitted to the national rural environmental cadaster (CAR) by December 2016<br>*Source*: System for Rural Environmental Cadaster (SICAR)<br>http://www.car.gov.br/publico/imoveis/index | We were not able to track the shift from public to private and neither the evolution of registration of existing properties. |
| **(#4)**<br>Undesignated public lands | (MT)<br>1,978.8<br>(5.6%)<br><br>(PA)<br>12,959.2<br>(27.9%) | State property regimes in which lands have no designated use. In principle, all social actors are excluded from use. | Non-regulated uses.<br>With weak governance, land and associated resources (e.g., commercial timber) are under dispute by social actors seeking to exclude each other from land use privileges.<br>Typically, irregular claimant users will not observe land use limits established by law and are less responsive to policies that require formal ownership, leading to resource depletion. | Polygons of Public Undesignated Forests<br>Source: Brazilian Forestry Service (SFB)<br>http://mapas.mma.gov.br/ | Undesignated lands shrank and gave place to settlements and strict use areas in<br>MT<br>**1985** 4,199.7 **2007** 2,240.5<br>**1995** 3,044.1 **2012** 1,978.8<br>**1998** 2,524.1<br>and PA<br>**1985** 35,134.3 **2006** 14,501.2<br>**1996** 34,696.9 **2008** 12,959.2<br>**1998** 32,729.2 **2012** 12,959.2<br>**2005** 28,098.6 |
| **(#5)**<br>Unknown tenure | (MT)<br>8,360.3<br>(23.7%)<br><br>(PA)<br>2,625.3<br>(5.7%) | Tenure systems may or may not have been defined but are not declared in publicly available datasets. If land is a private property, the required Legal Reserve equals to forested area by 2008 if the property is a smallholding. If not, restoration requirements will vary as described by the BFC. | [Permitted, when registered and committed to compliance to the BFC]<br>Cattle ranching, agriculture (large and small scale), Sustainable uses (timber and non-timber forest products extraction); low-to-high intensity systems.<br>Landowners/holders often don't comply to the BFC. | Defined by the absence of tenure allocation and ownership information. | Unable to track |

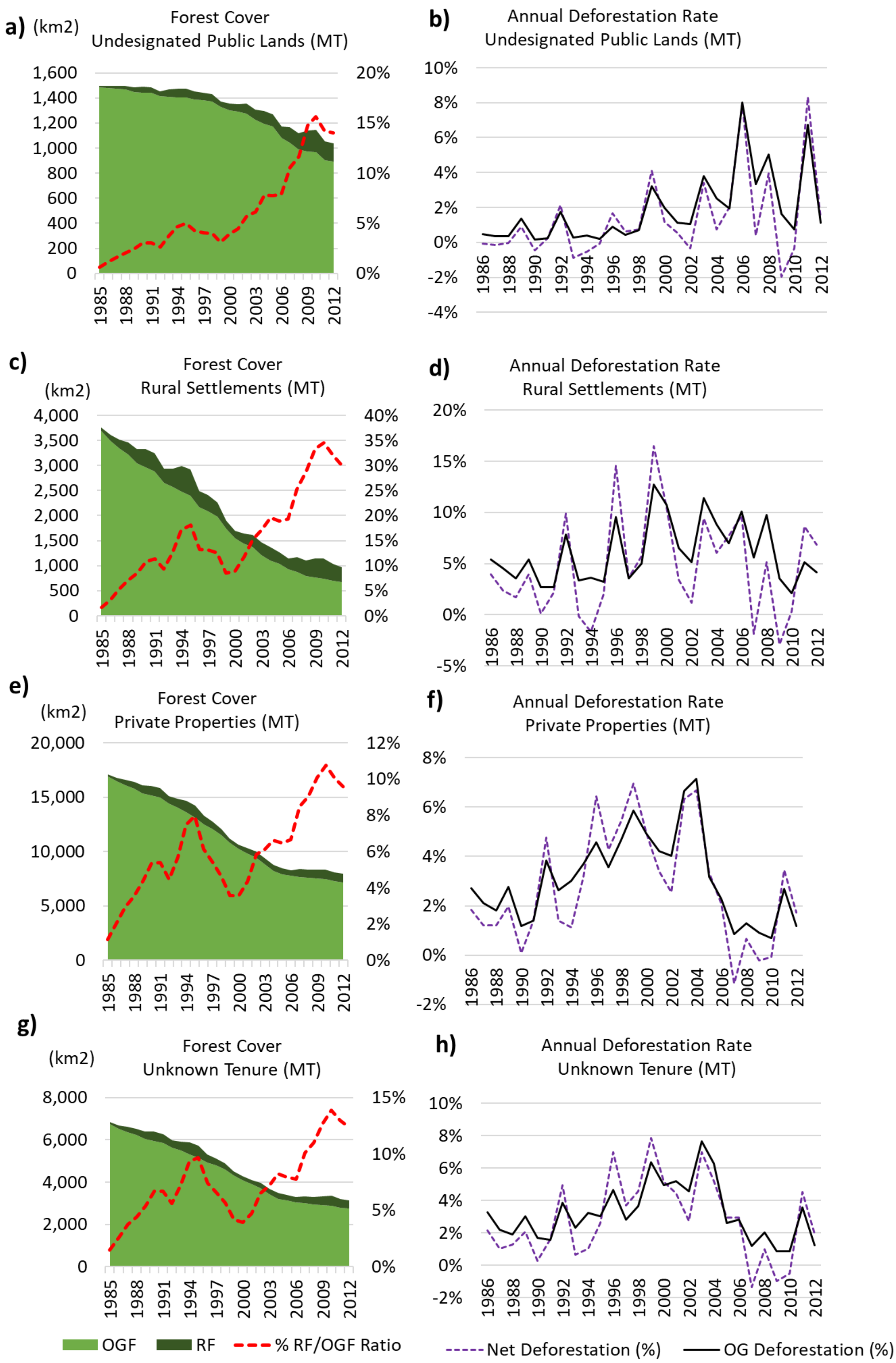
* Percentages refer to the share of the respective state overlapping the BR-163.

**Supplementary Table II-2. Carbon bookkeeping model parameters.**

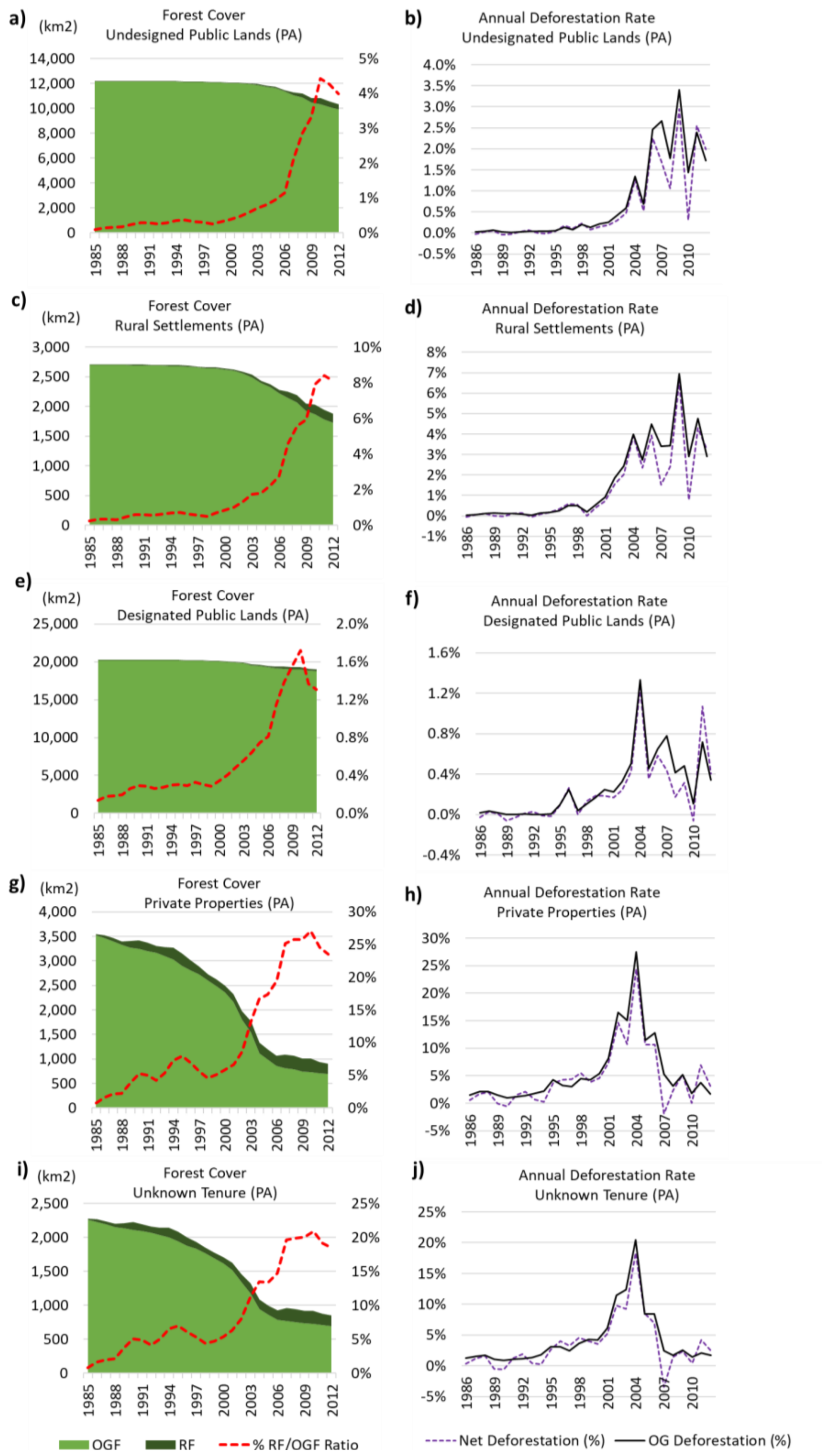
| Model Components | | Value/Unit | Spatial Dimension | Source |
|---|---|---|---|---|
| **Carbon fraction in Biomass** | - | 0.48 | Non-spatial (same rate across space) | Aguiar et al. 2012 |
| **Deforestation module parameters** | Deforestation | ≥1 hectare | Spatially Explicit | Müller et al. 2016 |
| | Fraction burned first year<br>Slash carbon fraction<br>Wood products carbon<br>Elemental carbon fraction<br>Reburn cycle<br>Annual decay rate wood carbon<br>Annual decay rate slash carbon<br>Annual decay rate elemental carbon<br>Annual decay rate BGB carbon | 42 %<br>15 %<br>41 %<br>2 %<br>3 (years)<br>10%<br>40%<br>0.1%<br>70% | Non-spatial (same rate across space) | Aguiar et al. 2012 |
| **Regrowing forests parameters** | Regrowth and Re-clearance | ≥1 hectare | Spatially Explicit | Müller et al. 2016 |
| | Annual Biomass Accumulation Rate (BRF) | 1.2% ((AGB+BGB) *0.012) | Fixed fraction of Total Biomass | Lennox et al. 2018 |
| | Annual Carbon decay rate from regrowth re-clearance | 100% | Fixed fraction of BRF | - |
| **Aboveground biomass (AGB)** | | 8-320 | Spatially Explicit | Leite et al 2012 |
| **Belowground biomass (BGB)** | | (AGB*0.3) | Fixed fraction of AGB | - |

**Supplementary Table II-3. Summary of Forest Cover Change (FCC) and Emission outputs.** OGF=Old-growth Forest; RF=Regrowing Forests

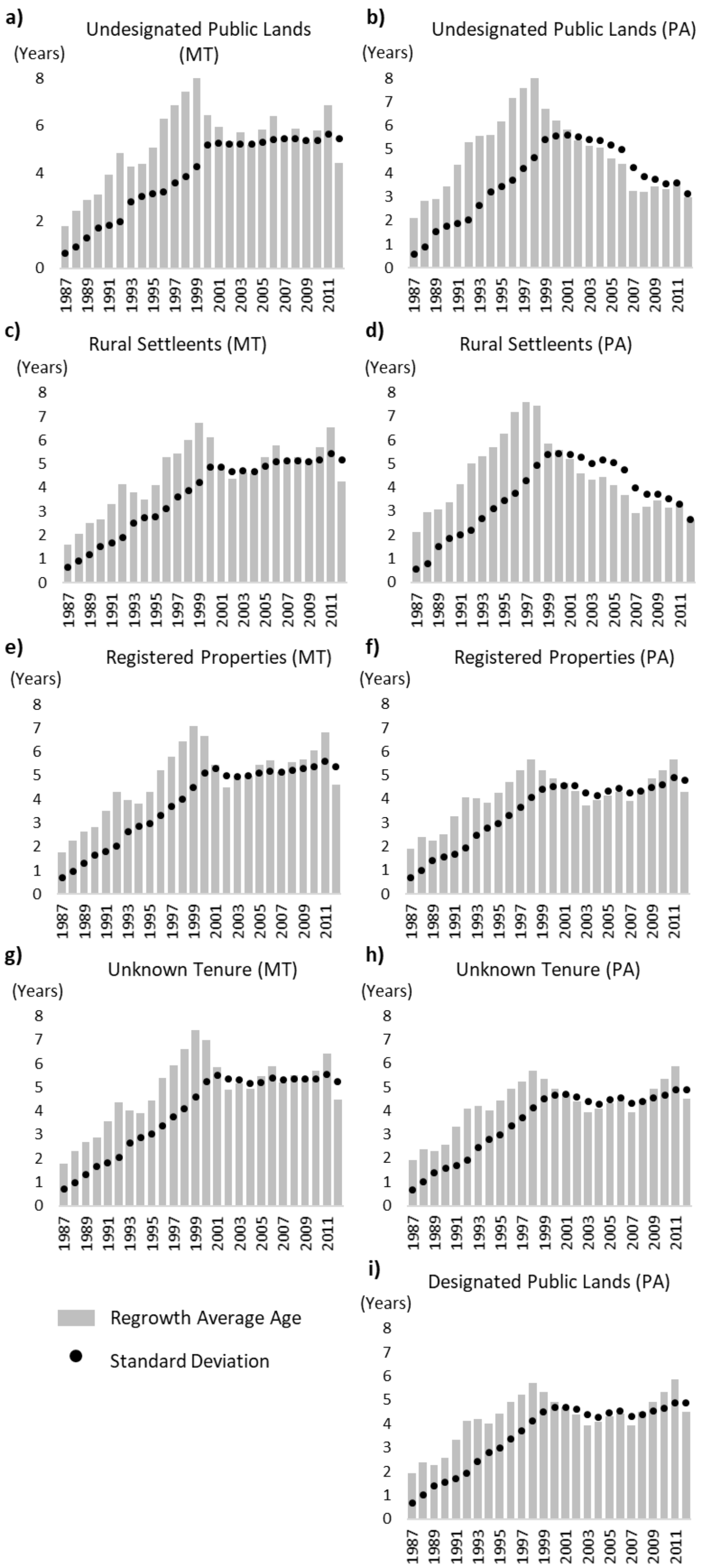
| | Mato Grosso | | | | | | Pará | | | | | | Total |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Und. Public Lands** | **Rural Settlements** | **Designated Public Lands** | **Private Properties** | **Unknown Tenure** | **Total (MT)** | **Und. Public Lands** | **Rural Settlements** | **Designated Public Lands** | **Private Properties** | **Unknown Tenure** | **Total (PA)** | **Total (BR-163)** |
| *Area ($km^2$)* | | | | | | | | | | | | | |
| Total | 1,978.8 | 4,565.8 | 42.6 | 20,294.1 | 8,360.3 | 35,241.0 | 12,959.2 | 2,780.0 | 24,184.8 | 3,879.4 | 2,625.3 | 46,428.7 | 81,670.2 |
| Non-Forest | 421.3 | 79.4 | 26.2 | 550.8 | 315.3 | 1,393.0 | 739.4 | 46.0 | 3836.7 | 73.0 | 178.1 | 4,873.3 | 6,266.3 |
| OGF (1985) | 1,484.0 | 3,691.6 | 15.9 | 16,898.4 | 6,728.2 | 28,818.0 | 12,156.2 | 2,703.0 | 20,229.3 | 3,518.4 | 2,257.1 | 40,864.1 | 69,682.1 |
| RF (1985) | 9.4 | 62.0 | 0.1 | 198.3 | 102.5 | 372.3 | 11.7 | 6.7 | 27.4 | 27.2 | 17.5 | 90.5 | 462.7 |
| Total Forest (1985) | 1,493.4 | 3,753.6 | 15.9 | 17,096.7 | 6,830.7 | 29,190.3 | 12,168.0 | 2,709.7 | 20,256.7 | 3,545.6 | 2,274.6 | 40,956.6 | 70,144.8 |
| OGF (2012) | 891.9 | 677.0 | 15.3 | 7,177.9 | 2,753.9 | 11,516.0 | 9,903.9 | 1,730.2 | 18,793.7 | 693.3 | 696.3 | 31,817.4 | 43,333.4 |
| RF (2012) | 145.2 | 291.4 | 0.2 | 759.4 | 388.0 | 1,584.5 | 411.9 | 152.0 | 249.5 | 213.0 | 157.7 | 1,184.0 | 2,762.8 |
| Total Forest (2012) | 1,037.1 | 968.4 | 15.5 | 7,937.3 | 3,141.8 | 13,100.0 | 10,315.8 | 1,882.2 | 19,043.2 | 906.3 | 854.0 | 33,001.5 | 46,096.2 |
| OGF Change (1985-2012) | 592.1 | 3,014.6 | 0.5 | 9,720.5 | 3,974.4 | 17,302.1 | 2,252.4 | 972.8 | 1,435.6 | 2,825.1 | 1,560.8 | 9046.7 | 26,348.7 |
| *(%) OGF Change (1985-2012)* | *-39.9* | *-81.7* | *-3.4* | *-57.5* | *-59.1* | *-60.0* | *-18.5* | *-36.0* | *-7.1* | *-80.3* | *-69.2* | *-22.1* | *-37.8* |
| RF Change (1985-2012) | 135.8 | 229.4 | 0.1 | 561.0 | 285.5 | 1,211.9 | 400.2 | 145.3 | 222.1 | 185.8 | 140.2 | 1,093.6 | 2,305.5 |
| *(%) RF Change (1985-2012)* | *+1,448.0* | *+370.1* | *+225.0* | *+282.8* | *+278.5* | *+325.5* | *+3416.7* | *+2177.2* | *+810.4* | *+682.9* | *+802.9* | *+1,208.9* | *+498.2* |
| *(%) RF/Total Forest Share (2012)* | *14.0* | *30.1* | *1.4* | *9.6* | *12.3* | *12.1* | *4.0* | *8.1* | *1.3* | *23.5* | *18.5* | *3.6* | *6.0* |
| *(%) RF/Deforested Area Share (2012)* | *21.9* | *7.7* | *19.7* | *6.0* | *7.3* | *7.1* | *17.8* | *15.1* | *16.2* | *6.8* | *9.1* | *12.2* | *8.6* |
| *Emissions (Tg $CO_2e$)* | | | | | | | | | | | | | |
| OGF Deforestation (by 2012) | 31.1 | 177.3 | 0.0* | 569.6 | 233.5 | 1,011.6 | 121.1 | 54.0 | 80.4 | 177.8 | 96.6 | 529.8 | 1,541.4 |
| RF deforestation (by 2012) | 1.1 | 4.9 | 0.0* | 11.1 | 5.4 | 22.7 | 1.6 | 0.7 | 1.6 | 3.2 | 2.0 | 9.2 | 31.7 |
| Uptake by RF (by 2012) | 1.5 | 5.8 | 0.0 | 13.6 | 6.7 | 27.7 | 2.6 | 1.0 | 2.4 | 4.0 | 2.6 | 12.6 | 40.3 |
| Net Emissions (by 2012) | 30.6 | 176.4 | 0.0 | 567.0 | 232.3 | 1,006.3 | 120.1 | 53.7 | 79.6 | 177.0 | 96.0 | 526.4 | 1,532.7 |
| *(%) Offset by RF uptake* | *4.7* | *3.2* | *10.8* | *2.3* | *2.8* | *2.7* | *2.1* | *1.9* | *3.0* | *2.2* | *2.6* | *2.3* | *2.6* |
| Mean Annual Net Emissions (1995-2004) | 1.0 | 7.7 | 0.0* | 29.2 | 11.5 | 49.4 | 1.6 | 1.2 | 2.5 | 9.4 | 5.1 | 19.8 | 69.2 |
| Mean Annual Net Emissions (2005-2012) | 2.1 | 4.0 | 0.0* | 12.3 | 5.2 | 23.6 | 12.7 | 5.1 | 6.6 | 6.7 | 3.8 | 34.9 | 58.5 |
| *(%) Change (1996-2005)/ (2006-2012)* | *+117.6* | *-48.0* | *+52.3* | *-57.8* | *-55.0* | *-52.2* | *+682.7* | *+319.1* | *+163.3* | *-28.4* | *-25.7* | *+76.1* | +15.3 |




**Supplementary Figure II-1.** (a,c,e,g) **Old-growth forest cover (km²)** (OGF; light green), **regrowing forest cover (km²)** (RF; dark green) **and ratio between regrowing and old growth forests** (%RF/OGF Ratio; red dashed line) (1985-2012) for Mato Grosso (MT); (b,d,f,h) **annual net deforestation rate** (%; representing the rate of total forest (old growth and regrowth) losses (+) or gains (-) relative to the previous year) and old-growth deforestation rate (%; representing the rate of old-growth forest losses (+) or gains (-) relative to the previous year) for Mato Grosso (MT).




**Supplementary Figure II-2.** (a,c,e,g,i) **Old-growth forest** (OGF; light green), **regrowing forest** (RF; dark green) **cover (km²) and ratio between regrowing and old growth forests** (%RF/OGF Ratio; red dashed line) (1985-2012) for Pará (PA); (b,d,f,h,j) **annual net deforestation rate** (%; representing the rate of total forest (old growth and regrowth) losses (+) or gains (-) relative to the previous year) and old-growth deforestation rate (%; representing the rate of old-growh forest losses (+) or gains (-) relative to the previous year) (1986-2012) for Pará (PA).




**Supplementary Figure II-3. Average permanence time ("age"; years)** of regrowing forests across tenure categories and states of Mato Grosso (MT) and Pará´(PA)

# Chapter III

# Regrowing forests contribution to law compliance and carbon storage in private properties of the Brazilian Amazon

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## Abstract

The viability of the climate pledges made by Brazil at the COP21 in Paris, 2015, heavily depends on the success of the country policies related to forest governance. Particularly, there are high expectations that the enforcement of the Brazilian Forest Code (BFC) will drive large-scale forest recovery and carbon mitigation. In this study, we quantified the potential role that ongoing forest regeneration may play in offsetting deficits from private properties with less vegetation cover than determined by the BFC, considering different law implementation settings. Focusing on the Amazon Biome, we overlaid property level data from a mandatory registry (≈ 250,000 properties) onto land cover maps to quantify on-site forest deficit offsets by ongoing forest recovery. Similarly, we estimated the share of regrowing forests in private properties potentially eligible for off-site deficit compensation (i.e., via market-based forest certificates trade). Regrowing forests could reduce, on-site, 3.2 Mha of forests deficits, decreasing non-compliance from private properties by 35%. Likewise, forest certificates availability increased by 3.4 Mha when we included regrowing forests in the calculations. On the one hand, trading certificates issued from recovering forests may represent a low-cost strategy for compliance with the BFC, a pathway for achieving restoration targets, and an additional source of income for landholders. To meet this potential, it is necessary to better conceptualize second-growth forests, advancing the poor definitions presented by the BFC, and offer an operational basis for their protection. On the other hand, including regrowing forests' certificates in compensation schemes may further restrain the potential of the trading mechanism for conservation of unprotected old-growth forests and lead to positive net carbon emissions. We highlight that the BFC implementation must be carefully regulated to maximize synergies between compliance and forest resources conservation and enhancement.

# 1. Introduction

Forest conservation and forest restoration are key strategies for mitigating the impacts of deforestation on biodiversity, soil, water quality and carbon stocks depletion (Banks-Leite et al. 2014; Barlow et al. 2007; Martin et al. 2013; Zhao et al. 2013). Hence, the rise of a global agenda on forest restoration has motivated many countries to scale-up forest recovery initiatives (Chazdon et al. 2017). Recent examples are international conventions such as the Aichi Targets, setting a restoration target of at least 15% of degraded ecosystems (Jørgensen 2013), and the Bonn Challenge, targeting the restoration of 350 Mha of degraded and deforested lands globally by 2030 (Bonn Challenge 2017).

Chazdon et al. (2016b) estimated that, if left to regrow, in 40 years regrowing forests (RF) in Latin America could offset two decades of fossil fuels and industrial carbon dioxide emissions from the region. Brazil alone accounts for 75% of the carbon storage potential of young to medium age second-growth forests of all tropical Latin America (Chazdon et al. 2016b). Focusing on the Brazilian Amazon, Aguiar et al. (2016) demonstrated that RFs could turn the Amazon into a net carbon sink in a few years' time, if, in addition to continued deforestation reduction, second-growth forests are expanded and protected.

Aware of this mitigation opportunity, Brazil pledged to restore 12 Mha of forests by 2030 at the COP21 (Nationally Determined Contribution, NDC - ratified at the UNFCCC COP Paris 21) (Brasil 2015). However, achieving this target depends on the implementation of interrelated sectorial policies involving different stakeholders, as well as legislation and market developments (Brancalion et al. 2016b; Lazos-Chavero et al. 2016). Chief among policies are the recently revised Brazilian Native Vegetation Protection Law (Law N. 12651/2012), commonly referred to as Brazilian Forest Code (BFC) (Brasil 2012), and the National Plan for Native Vegetation Recovery – PLANAVEG, launched in 2017 (Brasil 2017a).

In 2014, an assessment mapped over 17 Mha of secondary vegetation (in this article used as a synonym of regrowing forests) in the Brazilian Amazon Biome, inside public and private lands (INPE 2014c). However, very often, RFs are a temporary component of the landscape, quickly reincorporated into productive land. Between 2008 and 2012, 25% of RF areas in the Amazon were re-cleared while their total cover increased, suggesting that the short-life of this land cover may impair its long-term potential for carbon mitigation (Aguiar et al. 2016). Underlying land use systems and heterogeneous actors strongly influence RFs trajectories (Lambin and Meyfroidt 2010). In the Amazon, RFs appeared as a transitional cover with the purpose of ecological restoration in traditional agricultural systems, as revegetation in abandoned, degraded pastures or as a by-product of agricultural intensification (Costa 2016). Therefore, RFs are multifaceted components of land use systems, and competition for land associated with the lack of specific legislation regulating their protection, threaten the persistence and co-benefits of forest recovery (Barbier et al. 2010; Carvalho et al. 2019a; Vieira et al. 2014).

The BFC regulates the conservation of native vegetation in private lands. In Brazil, 54% of the native vegetation is located inside private properties (Sparovek et al. 2015) making landholders' compliance with the BFC strategic for forest conservation and restoration. Recent studies have calculated forest deficits (i.e., forest cover falling behind with the BFC requirements) and forest extent apt for issuing certificates (which may be used for off-site deficit compensation) for rural properties (Brito 2017; Freitas et al. 2017b; Micol et al. 2013; Nunes et al. 2016; Soares-Filho et al. 2014). These studies indicated that BFC compliance levels (i.e., landholdings degree of agreement to legal conservation requirements) and forest balances (i.e., regionally aggregated difference between potential forest certificates availability and deficits) vary in intensity and spatial distribution.

The forest balance of a federal state is an indicator of its potential role as buyer or provider of forest certificates eligible for trade via a market-based mechanism for BFC deficits offsetting. In this context, state-level regulatory setups are crucial (Freitas et al. 2017b); they impose different restrictions for trade, delimiting the size and appeal of the market, with consequences for economic gains and conservation additionality. Currently, the implications of using different offsetting mechanisms foreseen by the BFC are under evaluation by state governments and sectors of the civil society (especially by the academia and NGOs), with regards to market territorial restrictions, protection status of the traded forest certificates and prioritization of vulnerable areas (Freitas et al. 2017b; Gasparinetti and Vilela 2018; Soares-Filho et al. 2016). However, less attention has been given to the eligibility and potential contribution of adding RFs to the BFC balance and the consequences for conservation. Brito (2017) found that in the state of Pará, 30% of forests apt for issuing tradable certificates likely come from regrowing forest areas, indicating the need to investigate possible implications for related policies.

In this paper, we bring this discussion forward and investigate the potential contribution of RFs to law compliance and conservation under the BFC in the Brazilian Amazon. Our specific goals are: (1) To assess the current compliance with the BFC on property-level, including and excluding RFs of landholdings' forest stocks; (2) to evaluate the implications of alternative regulatory setups of BFC implementation, including and excluding RFs, for forest conservation and carbon storage. The paper is organized as follows. First, we detail central aspects of the BFC to contextualize our analysis. Then we present the methods for achieving goals 1 and 2, followed by the results and discussion of our findings under the perspective of our guiding questions (see next section) and previous research on the topic. We also discuss the challenges and implications of implementing the proposed setups in the context of the Brazilian Amazon and make suggestions for future work.

## 1.1. Pathways to compliance with the Brazilian Forest Code: including forest regrowth

In 1965 the BFC instituted two conservation categories inside private lands, the Legal Reserve (LR) and the Permanent Protection Area (PPA) (Supplementary Table III-1) (BRAZIL, 1965). The PPAs define strict protection zones inside the properties (i.e., riparian buffer zones and steep terrain). The LR is a set-aside for native vegetation protection, defined as 80% of the landholding in forestlands of the Amazon biome (Supplementary Table III-2) since an addendum made to the BFC in 1996 (Law MP 1.511/1996). However, the BFC was never properly enforced, which, in combination with conflicting land governance, led to massive noncompliance among landholders (Sparovek et al. 2012). The ineffectiveness of the law motivated its revision, aiming to create instruments to give noncompliant farmers access to laxed conditions to regularize their situation – a lengthy and controversial process that mobilized the civil society and was marked by conflicts between conservationists and the agribusiness sector. The revised BFC, promulgated in 2012 (Brasil 2012), weakened restoration requirements for noncompliant landholders and granted amnesty to most irregular deforestation prior to 2008 (Soares-Filho et al. 2014).

The 2012 BFC created two sets of rules, not mutually exclusive, to address conservation and compliance in private properties (Supplementary Table III-2). The first regulates the conservation of forest remnants. The second specifies the minimum requirements noncompliant landholders must conform to become law-abiding. As a first step, landholders must submit georeferenced information of property boundaries to a countrywide land registry (i.e., Environmental Rural Registry - CAR, Portuguese acronym) planned to support the BFC monitoring and enforcement. Next, the roadmap to compliance should be detailed in a 20 years length plan for environmental regulation (i.e., PRADA, acronym in Portuguese) (Supplementary Table III-1) (Brancalion et al. 2016a) with strategies including on-site (i.e., forest recovery) or off-site compensation (Oakleaf et al. 2017). The PRADA must conform to the Program for Environmental Regulation (Supplementary Table III-1; i.e., PRA, Portuguese acronym), a state level legislation guiding the application of the BFC, ideally tailored to maximize law compliance and lateral benefits of the law implementation in each federal state (Supplementary Table III-3). If regulated and enforced, the BFC may promote on-site forest recovery through either native or mixed species forest reestablishment, and off-site forest recovery or forest conservation when the compensation pathway is chosen for compliance. Areas deforested before 2008 could either be recovered on-site or compensated off-site, but on-site forest reestablishment is mandatory for areas deforested after 2008 (Brasil 2012).

Although off-site compensation precedes the 2012 BFC (e.g., prior to 2012, noncompliant farmers could acquire properties with exceeding forest to solve their deficits), the current version of the BFC institutionalized tradable certificates of private protected and unprotected forests framed within Environmental Reserve Quotas (Supplementary Table III-1; i.e., CRA, Portuguese acronym), that is, eligible to be used for off-site compensation. The CRA mechanism was created to be a cost-effective strategy for deficit compensation through the acquisition of forest certificates based on predefined duration contracts. Landholders can indistinctively use old-growth and secondary vegetation, at any stage of recovery, to compose their properties' LRs (Article 46, Item I of the Law N. 12767/2012). A priori, tradable certificates may also be issued from regrowing vegetation areas, unless state PRAs explicitly oppose, as it is the example of Mato Grosso do Sul (Mato Grosso do Sul State Decree 14,722 of 2015).

In 2017 a federal plan named PLANAVEG was launched (MMA 2017b), with the aim of enabling the 12 Mha restoration target under the terms of the BFC and aligned with the Brazilian NDC (Celentano et al. 2017). Policy implementation is strategic for achieving the NDC's targets, as Brazil's pledges are not conditioned to international funding. It will be necessary to set a baseline for old-growth and regrowing forests on private rural landholdings from which to estimate the policies' contribution to forest expansion. This information is crucial to guide the regulation and execution of mitigation plans based on the BFC.

Yet, despite their promising role, RFs are poorly addressed by the BFC and other pieces of environmental legislation in Brazil (Vieira et al. 2014). The BFC does not provide a definition of regrowing forests, nor clarifies how to monitor and enforce their protection. In addition, state-level regulations make superficial mentions to recovering forests, their definition and protection status and eligibility to compose LRs or to be used as compensation assets (Supplementary Table III-3). No technical guidance is provided on how to identify such regrowing forests (e.g., temporal or biophysical criteria), except for the state of Pará (State Law IN-08 of 2015). As legal definitions are vague, landholders may fear ambiguous interpretations of the BFC concerning the use and potential protection of RFs for compliance with the BFC or show resistance to engage in new alternatives for compliance (Pacheco et al. 2017).

As a first step to include forest regrowth in the discussion about the BFC implementation, it is important to understand where RFs may contribute the most to the BFC compliance and how it may interfere with the offer and demand for forest certificates, either increasing competition with unprotected old-growth forest (OGF) surplus or as a compensation choice in states with limited certificate offer. Dependent on the CRA trade regulatory settings, the increased offer of certificates issued from regrowing forest areas may decrease the appeal of the CRA market for certificates issued from unprotected old-growth – carbon rich – forest areas. Therefore, we designed this analysis to consider plausible outcomes of different regulatory settings, implementable by the states PRAs, to better understand the potential role of RFs for law compliance, forest conservation and carbon storage. The guiding research questions were:

(1) How does the inclusion of RFs in LRs changes the BFC forest balance? How much LR deficit can be offset on-site, by the inclusion of ongoing regeneration in LRs demarcation?

(2) How much do certificates issued from OGF and RFs contribute to BFC compliance under different regulatory setups of CRA market restriction, excluding or including ongoing regeneration from LRs and from forest certificates?

(3) How much forest carbon could be offset and protected under different regulatory setups of CRA market restriction, excluding or including ongoing regeneration from LRs and from forest certificates?

# 2. Material and methods

## 2.1. Study area

We focused on private rural properties in the Amazon Biome which intersects the nine federal states composing the Brazilian Legal Amazon (BLA) (Figure III-1a). Most of the study area is covered by humid tropical forest, but significant portions are covered by savannas or grasslands (Figure III-1b – Non-Forest class). Forest loss advances from east to west and south to north, along major roads, concentrated in the states of Pará (PA), Mato Grosso (MT) and Rondônia (RO). Eighty percent of the original forest cover is standing, and 22% of the cleared area is covered by secondary vegetation with different levels of recovery (INPE 2014b; INPE 2014c).

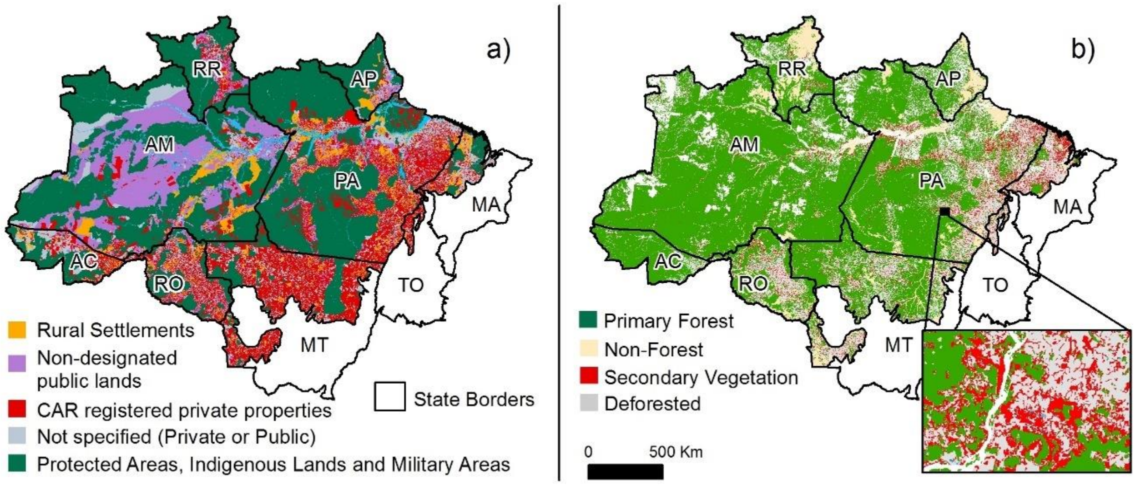



**Figure III-1. Study area.** (a) Land Categories and (b) Forest Cover, old-growth and secondary– detailed view showing secondary vegetation spatial patterns; AC=Acre; AM=Amazonas; AP=Amapá; MA=Maranhão; MT=Mato Grosso; PA=Pará; RO=Rondônia; RR=Roraima; TO=Tocantins. Data sources: FUNAI (2017), MMA (2017d), INCRA (2017), INPE (2014b); INPE (2014c)

## 2.2. Estimating forest deficits and forests apt for issuing certificates at property level

We quantified forest deficits and forest cover eligible for issuing certificates (i.e., CRAs) in rural properties, based on the conservation and regularization (compliance) requirements established by the revised BFC (Brasil 2012) (Supplementary Table III-2). For each property, we divided forest stocks in three categories (a) LR non-eligible for issuing CRAs, (b) LR eligible for issuing CRAs (i.e., protected private forest), and (c) forest surplus exceeding the 80% LR requirements, eligible for issuing CRAs or for conversion to other land uses (i.e., unprotected forest surplus) (see Supplementary Table III-1 for definitions of forest stocks categories). We also identified and divided forest shortfall (deficit) in two categories: (d) LR deficit qualified for on-site or off-site offsetting, and (e) post-2008 deforestation, where on-site forest reestablishment is mandatory (Supplementary Table III-2, Supplementary Figure III-1). First, we estimated the five categories (a-e) mentioned above considering only OGF stocks inside properties. Next, we calculated the contribution of regrowing forests to the BFC balance including RFs areas in forest stocks. From that we estimated how much of the area deforested after 2008 is currently regrowing and how much additional on-site forest regeneration is required to achieve LRs compliance with the BFC.

LRs are placed by landowners at their determination, contingent on government approval. However, for this study, we mapped LRs based on forest extent inside properties, identified by overlaying land cover maps with property boundaries. We did not distinguish between forests located in LRs and in PPAs, meaning all forest stocks add up to LRs, which is aligned with Art. 15 of the BFC that made admissible the inclusion of PPAs in LRs to reduce landholders' deficits. We did not calculate PPAs deficits, which have a specific location at environmentally fragile areas inside properties as they could not always be captured by the spatial resolution used in our study (e.g., riparian protected areas may be wide as 5 meters, while our analysis spatial resolution is of 100 meters).

We applied a collection of spatial datasets, resampled to a 100-meters resolution, including individual property boundaries, forest and land use cover with 2014 as a base-year. We downloaded individual rural properties included in the CAR (SICAR 2017) before December 2016. Current regulations determine that all the landholders must register their properties before December 31st, 2019, to compose a provisional CAR. One important disadvantage of the CAR data is that, at this provisional stage, there are no impediments to the inclusion of false or conflicting information (e.g., overlapping properties, double registry). Therefore, after downloading the dataset, we removed inconsistencies that led to the reduction of properties included in the study from *n-Initial*=420,778 to *n-Final*=255,224 (Supplementary Figure III-1) covering 15% of the Amazon biome. We also collected information on protected areas, indigenous lands, and military areas – considered as public areas destined to conservation (Figure III-1a, Supplementary Table III-4), and consolidated areas for agriculture (FUNAI 2017; MMA 2017d). These were used to calculate the varying BFC conservation requirements for the individual properties depending on their location and the municipality forest protection level. (Supplementary Figure III-1; Supplementary Table III-1, Supplementary Table III-2 and Supplementary Table III-4).

To quantify OGF area per property, we used old-growth forest cover and deforestation data (Figure III-1b; Supplementary Table III-4) annually provided by the National Institute for Space Research (INPE) through its deforestation monitoring program (PRODES) since 2000 (INPE 2014b). We distinguished between deforestation occurring before and after 2008, the year established as a threshold to grant access to relaxed terms for compliance according to the BFC (e.g., amnesty for small landholders, access to compensation via forest certificates – CRA, see Supplementary Table III-2, "RL – Regularization Regime").

RFs cover was also obtained from INPE, through the TerraClass project (INPE 2014c). This project mapped post-deforestation land use and cover, including "secondary vegetation" and "pasture covered by forest regrowth" classes, for the Brazilian Amazon for five time-steps (e.g., 2004, 2008, 2010, 2012, and 2014). We used this information to derive RFs cover area and age (i.e., annual landscape permanence time of forest regrowth per pixel) in the Amazon. Supplementary Table III-4 details our dataset. We only considered forestlands, and excluded properties overlaying savannah or natural grasslands from our analysis, using the PRODES forest-non-forest mask as reference. This decision was made due to the absence of spatial information on land use and cover for vegetation types not included in the current monitoring systems (INPE 2014b; INPE 2014c).

We conducted one additional BFC balance analysis excluding secondary vegetation mapped with less than 5 years of prevalence in the landscape by 2014 To avoid the inclusion of fallow lands mapped as "secondary vegetation" by TerraClass. We used a 5-years temporal criterion based on studies which have identified that for some regions of the Amazon, the average permanence time of RFs in the landscape is 5 years (Aguiar et al. 2012; Müller et al. 2016b; Schwartz et al. 2017).

## 2.3. Designing alternative regulatory setups

To support the discussion on the impact of including RF certificates in a CRA market, we considered eight different regulatory setups (Figure III-2; see Supplementary Table III-5 for a description of the eight regulatory setups applied). This exercise allowed us to understand the impact of different regulations on compliance and conservation additionality under market implementation. These setups combine three variables: (a) the eligibility of certificates issued from RFs; (b) the spatial coverage of the market (i.e., either restricted to federal state boundaries or open for trade within the biome); and, (c) the protection status of forest certificates (i.e., from protected private forests or from unprotected forest surplus).

In a self-regulating market the CRAs issued from unprotected private forests would have higher prices than those issued from protected private forests which, by definition, do not allow alternative land uses, hence have very low opportunity costs. Therefore, we assumed that CRAs issued from protected forests would be absorbed first by the market, in detriment of unprotected, more expensive CRAs. We used the following hierarchy to calculate the BFC balance for our market regulatory setups: old-growth protected forest → regrowing protected forest → old-growth unprotected forest → regrowing unprotected forest.

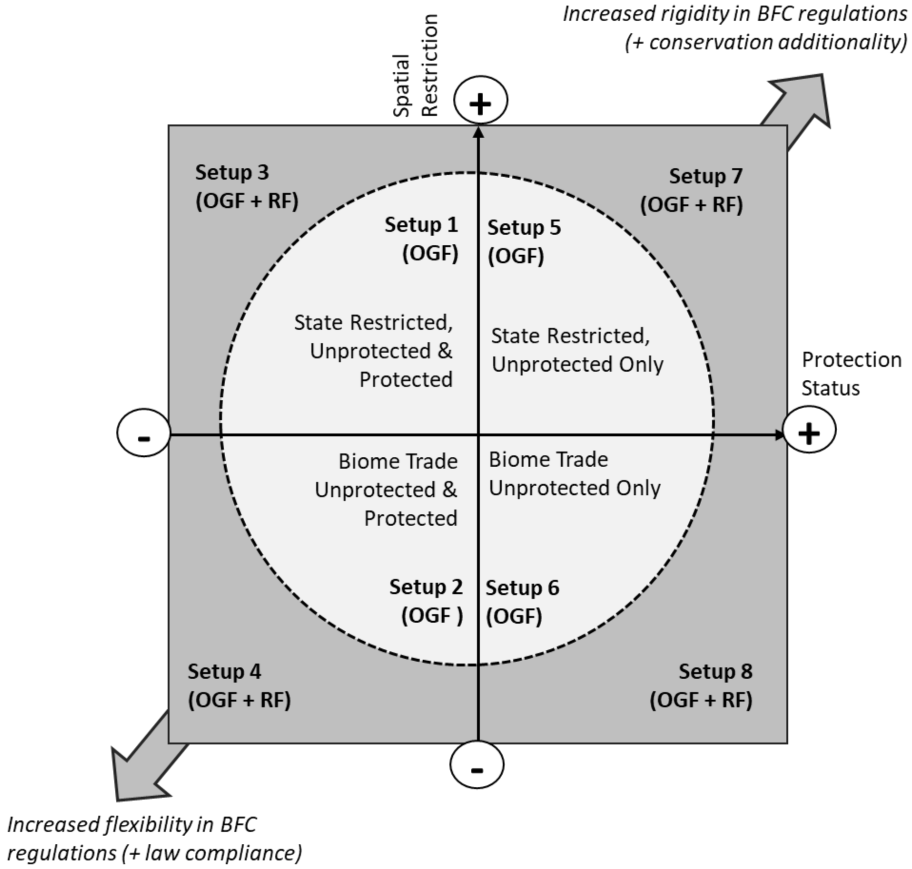



**Figure III-2.Combination of spatial and conservation criteria assessed by the regulatory setups for a forest certificate compensation mechanism.** *X* axis describes the protection status of forest apt for compensation: on the left, both protected and unprotected forests are allowed, on the right, only unprotected forests; *Y* axis describes the spatial coverage allowed for each regulatory setup: on the bottom, certificate trade is biome-wide, while on the top supply is restricted to state boundaries. The inner circle represents the settings excluding RFs. Large arrows point to the increase in either rigidity or flexibility in trade regulations with outcomes for law compliance and conservation additionality.

## 2.4. Carbon storage quantification and sensitivity analysis

We estimated current (actual) carbon stocks in OGFs and RFs and the potential carbon sequestration by RFs and forest deficits recovery; these values supported a discussion about the potential of the BFC for private forests carbon protection and sequestration under the implementation of the proposed market regulatory setups.

We used above and belowground biomass maps provided by the Third Brazilian Emissions Inventory (Brasil, 2016) as a reference to extract average forest carbon density (tons per hectare) for each landholding. The inventory maps provide original biomass values, expected to occur in undisturbed forests. The biomass estimates are based on a large compilation of plot level and literature data made spatially explicit using geostatistical methods. Other carbon pools such as understory, fine litter and soil carbon were not included.

Total carbon stocks were estimated for each of the private forest categories (i.e., protected forest not eligible for issuing CRAs, protected forest eligible for issuing CRAs and unprotected forests) -, and deficits (i.e., deficits offsetable on-site or off-site and deficits from post-2008 deforestation). To calculate total carbon stocks in OGFs and the potential carbon sequestration by RFs and deficit restoration we multiplied the biomass density by the forest cover or deficit extent. We transformed biomass values to carbon content using a conversion factor of 0.5. Current carbon stocks in RFs were obtained as described by equation 1, where pristine "carbon" density values were multiplied by an annual biomass accumulation rate "R" of 1.2% (Lennox et al. 2018) and by RFs age "i" values obtained by overlaying the bi-annual Terra Class maps.

$$\sum_{i=1}^{n} Carbon * R * Age_i \qquad \text{Equation 1}$$

Following, we estimate carbon stocks protection and sequestration potential under the different regulations of the CRA market. CRA units are measured in hectares being equivalent on their offsetting purpose but may differ in terms of carbon storage potential. To address this source of uncertainty, we conducted a sensitivity analysis of carbon storage potential of CRAs. We performed 1,000 random selections of properties with available certificates adding up to the area required to offset LR deficits in each regulatory setup and reported the mean, maximum and minimum respective carbon stocks from the combinations. The same was done for estimating the carbon sequestration potential by properties restoring remaining LR deficits if, according to the regulatory setup, the CRA demand is not covered by certificates availability. Finally, we evaluated the net carbon stocks protection resulting from each of the eight regulatory setups considering two additionality baselines. Baseline 1 considers the demand for certificates issued from unprotected forests as BFC additionality. Baseline 2 considers the demand for certificates issued from unprotected forests, the carbon sequestration from the remaining LR deficits and any protection of regrowing forests as BFC additionality. Results were expressed in carbon dioxide ($CO_2e$) values.

# 3. Results

## 3.1. BFC balance at property level excluding and including regrowing forests

We estimated that accounting for regrowing forests reduced, on-site, 3.2 Mha of LRs forest deficits. This represents a 35% decrease in offset requirements for private properties analyzed by this study (Figure III-3). Likewise, protected LR eligible for issuing CRAs increased by 3.4 Mha when we included RFs in calculations, adding up to 12 Mha (Figure III-3; Table III-1). Finally, including RF areas increased by 0.5 Mha the share of unprotected forests - eligible for legal deforestation in private properties (e.g., located in properties that have over 80% of forest cover and no deforestation since 2008). Most vegetation cover is not apt for compensation under the BFC (24.5 Mha considering only OGFs, and 27.5 Mha considering OGFs and RF areas) (Table III-1; Figure III-3). By 2014, one quarter of areas deforested after 2008 were regrowing vegetation (Supplementary Figure III-2b) representing a 0.4 Mha reduction in deficits that are non-eligible for offset via extra-property compensation. Restricting our analysis to RFs with a permanence time equal to or higher than 5 years (Supplementary Figure III-2a) noticeably decreased RFs contribution to BFC compliance (Table III-1; Figure III-3).

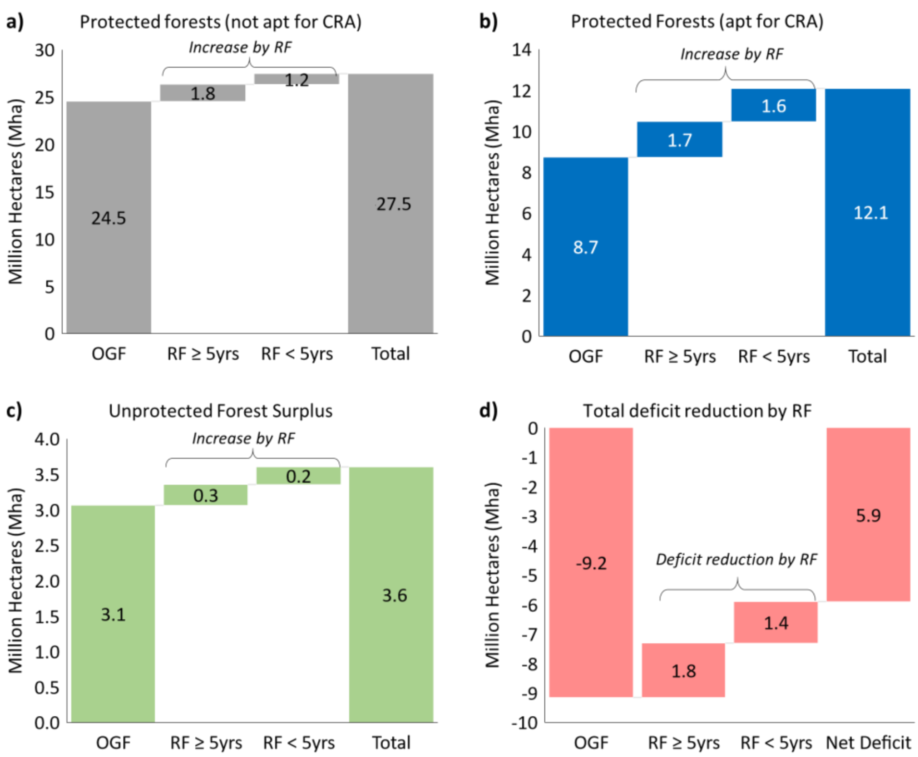



**Figure III-3. Increase in area by forest category** (a-c) and deficit reduction (d) after including regrowing forests in the forest balance calculations; (d) includes both LR deficits and post-2008 deforestation deficits.

**Table III-1. Legal reserve (LR), forest categories and deficits classification in million hectares** (Mha). (OGF) refers to calculations excluding regrowth; (RF) refers to the contribution of regrowing forests; (RF≥5yr) refers to the contribution of regrowing forests with permanence time equal or above 5 years.

| Forest | **Class** | **OGF** | **RF** | **RF≥5yr** | **Total (OGF+RF)** | **Total (OGF+RF>5yr)** |
|---|---|---|---|---|---|---|
| **Forest Categories** | Protected forests (not apt for CRA) | 24.5 | 3.0 | 1.8 | 27.5 | 26.3 |
| | Protected forests (apt for CRA) | 8.7 | 3.4 | 1.7 | 12.1 | 10.4 |
| | ***Sub-Total (Protected)*** | ***33.2*** | ***6.0*** | ***3.5*** | ***39.6*** | ***36.8*** |
| | Unprotected forest surplus | 3.1 | 0.6 | 0.3 | 3.6 | 3.4 |
| | **Total** | **36.3** | **6.9** | **3.8** | **43.2** | **40.2** |
| **Deficit** | LR deficits (apt for off-site compensation) | 7.5 | | | 4.6 | 5.7 |
| | Post-2008 deforestation deficits | 1.7 | | | 1.3 | 1.6 |
| | Regrowing in post-2008 deforestation areas | | 0.4 | 0.1 | | |

OGFs (protected and unprotected) apt for CRA surpass LR deficits in six states (AC, AM, AP, PA, RO, RR); in these states, LR deficit could be totally offset via certificate trade within state borders, without additional certificates issued from RF areas (Figure III-4a). For MT, MA and TO, LR deficits exceed the availability of protected and unprotected CRAs, and regularization will require forest recovery or the acquisition of forest certificates outside state borders. If included, certificates issued from RFs could turn the forest balance positive in MT but would not suffice in TO and MA (Figure III-4b). RFs are mainly concentrated in states with higher forest deficits (MT, MA, PA and RO), and a large deficit reduction could be achieved by conserving RFs areas as LR. Despite having a positive balance, PA has the second largest on-site forest deficit in the Amazon and could increase law compliance by 43% with the inclusion of RFs in LR (1.46 Mha). Deficits reduction were also high in in RO, MT, MA and TO, in relative and absolute numbers. Among the states with less deficits, including RFs in the forest balance calculation substantially reduced the percentage of deficits for RR, AM and AP (Figure III-4a-b).

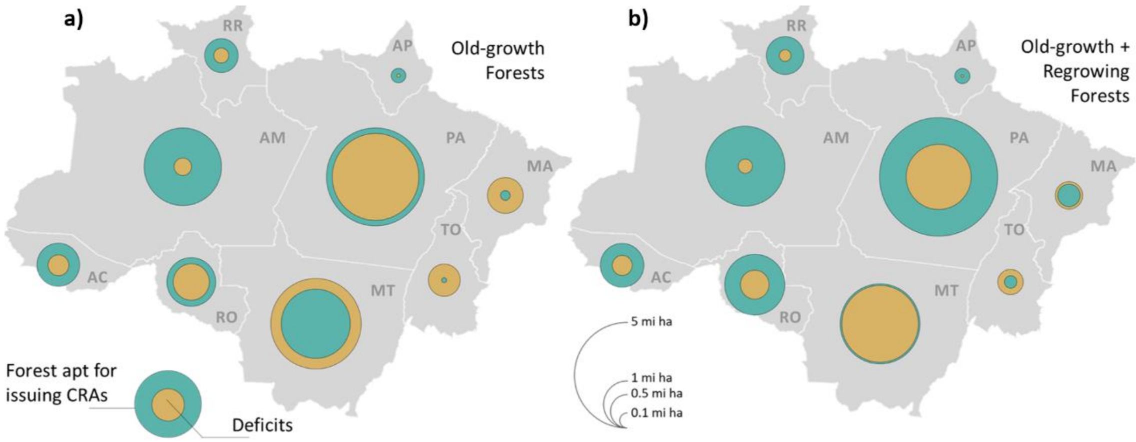



**Figure III-4. Forest balance for the analyzed properties per state.** The figure does not include forests not apt for issuing certificates. Mustard areas encircling blue areas means the total deficit is higher than the availability of forests apt for CRAs, and the opposite means that apt forest area is higher. Circle sizes indicate absolute area in Mha; (a) forest balance including OGFs only; (b) forest balance including OGFs and RF areas by 2014.

## 3.2. Regulatory setups results including and excluding regrowing forests

Figure III-5 illustrates the outcomes of different market regulatory setups (Figure III-2, Supplementary Table III-5) for law compliance and conservation additionality. As a rule, a less-restricted market shifted the balance between supply and demand towards certificate excess. Regulatory setup 3 included OGFs and RFs and hence started with less deficits to offset (4.4 Mha) than regulatory setup 1 (7.1 Mha) meeting most demand for certificates (98%) - even being state constrained -, in comparison to its equivalent (setup 1) which excluded RFs (75%). Regulatory setups 1 and 3 offered some additionality and absorbed 0.8 Mha and 0.5 Mha of unprotected forest certificates, respectively. Regulatory setups 2 and 4 also included protected forests apt for CRAs and did not impose any geographical constraint (i.e., offsetting allowed across the biome). In both setups CRA demand was fully met, indicating that the inclusion of apt protected forests in combination with a biome-wide forest trading scheme counteracts conservation additionality regardless of the inclusion of RFs.

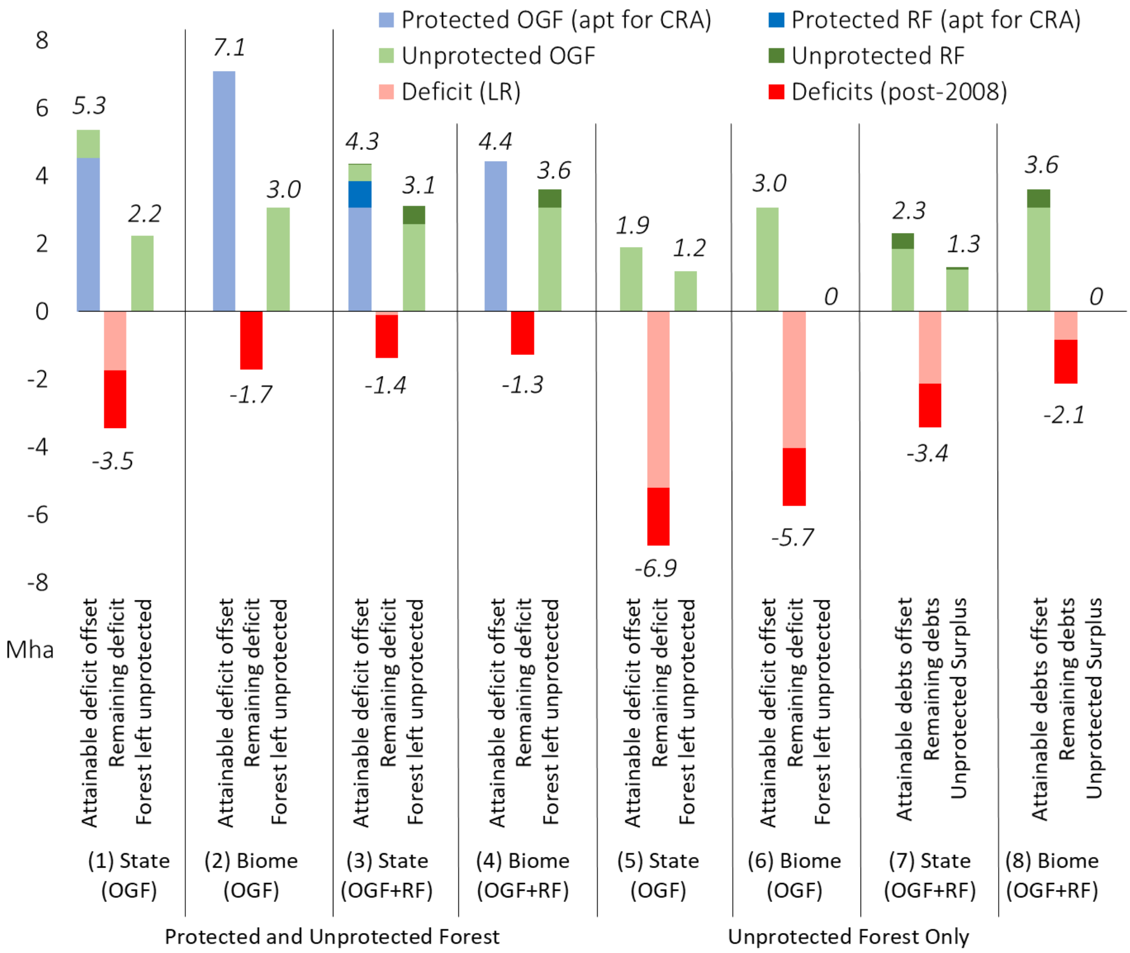



**Figure III-5. Effects of tested market regulations on law compliance.** *Attainable deficit offset* shows the maximum deficit reduction that can be expected within each setup, with colors highlighting forest stock categories that contributed to compensation. *Remaining deficits* are LR forest shortages that could not be offset given the market regulations. *Forest left unprotected* equals the amount of forest surplus that would not be assimilated by the market being left unprotected (see Supplementary Table III-2 for forest stock categories). As regulatory setups 1-2, 5-6 exclude on-site regeneration from LRs, the initial demand for forest certificates totaled 7.1 Mha. Regulatory setups 3-4 and 7-8 include RFs in LRs, which resulted in a lower demand for forest certificates (4.4Mha). (OGF) refers to setups in which only old-growth forests were considered in the BFC balance calculations; (OGF+RF) refers to setups in which old-growth forests and RF were considered in the BFC balance calculations.

To prioritize additionality, regulatory setups 5 to 8 limit off-site deficit compensation to the acquisition of certificates issued from unprotected forest surplus. Despite limiting certificates to unprotected surplus, regulatory setup 5 is unable to absorb the full certificate supply (26% of deficit offset) and, after exhausting compensation possibilities, would require 6.9 Mha of on-site restoration of to achieve full compliance. In this case, opening the trade for the biome (setup 6) would benefit law compliance (69% of deficit offset) and create more opportunities for conservation of unprotected surplus. Finally, regulatory setups 7 and 8 show the potential for an increase in law compliance after the inclusion of RFs certificates. Again, if trade is restricted to certificates from unprotected surplus then a biome-wide market would be able to fully absorb the OGFs and RFs surplus.

### 3.3. Old-growth and regrowing forests carbon storage quantification

OGF hold the equivalent to 20.7 $PgCO_2e$, while, by 2014, RFs stored 0.2 $PgCO_2e$. If left to regrow, stocks of ongoing forest regrowth could reach 3.8 $PgCO_2e$ (Supplementary Table III-6). Most OGF carbon stocks are, in theory, protected by law (91.0%) and are associated with forests not apt (65.5%) or apt for issuing CRAs (25.5%); the remainder (9.0%) are associated with forests eligible for alternative uses (unprotected surplus). If governed by the BFC, RFs potential carbon would be split between non-eligible (41.4%), eligible for compensation (49.6%), and unprotected surplus forests (8.9%).

If the baseline to analyze forest conservation additionality is the increase of protection above the minimum BFC requirement (i.e., 80%), then only regulatory setups 6 and 8 would be able to fully protect the carbon stocks in forest surplus (Supplementary Table III-7). All other setups made private properties a net source of carbon, as unprotected forest surplus was not fully assimilated by a CRA market, being left vulnerable to deforestation (Supplementary Table III-8). However, if, in addition to forest surplus protection, we consider the attainable (potential) carbon stocks in protected RFs and the recovery of the remaining LR deficits as BFC enforcement additionality, then most regulatory setups would present a positive balance with scenario 8 leading to the net protection of 5.0 $PgCO_2e$ (Supplementary Table III-7). On the other extreme, for setups 1-4, not even the protection of current carbon stocks in RFs and the recovery of remaining LR deficits would be enough to compensate for the eventual deforestation of unprotected OGF, leading to a negative balance (Supplementary Table III-7). A full account of carbon storage for each regulatory setup and different categories for each federal state can be found in Supplementary Table III-8 and Supplementary Table III-9.

## 4. Discussion

In this paper we use property-level data to provide the first comprehensive overview of regrowing vegetation potential contribution to compliance with the BFC, contextualizing the socio-environmental relevance that RFs may gain under the BFC implementation. Over 0.4 Mha of forests cleared after 2008 are recovering and could offset 24.7% of the post-2008 deforestation deficit for selected properties (Table III-1). An effective implementation of the BFC, supported by the validation of the CAR, should allow the separation between regrowth taking place on properties eligible for off-site compensation and over post-2008 deforested areas, and enforce the protection of RFs where on-site recovery is mandatory. Additionally, if RFs had the same protection status as OGFs, the BFC enforcement could secure the conservation of 6.3 Mha of forests recovering on farms with forest area below conservation requirements (Table III-1). This amount exceeds the 4.8 Mha of forest expansion planned to take place in the Amazon (BRASIL, 2017), which emphasizes the huge potential that a careful regularization (and enforcement) of RFs protection may have for promoting large scale forest restoration. However, current federal and state legislations do not make clear if and how RFs located in properties with LRs below the 80% cap should be protected by the BFC (Supplementary Table III-3).

An excessively flexible CRA market may represent a missed opportunity for the protection of 1.9 PgCO2e stored in unprotected OGFs (Supplementary Table III-7). Our results show that the inclusion of RFs in LRs can decrease deficits on-site, which further combined with lenient CRA trade regulations has the potential to fully solve offsetable deficits at the expenses of unprotected OGF conservation (setups 2-4, Figure III-5, Supplementary Table III-7). This reinforces that regulations should be based on an understanding of the potential role a state may play as a supplier or buyer of CRA (Gasparinetti and Vilela 2018) and, ideally, prioritize deficit compensation with OGF surplus in detriment of protected or RF certificates. States with high CRA offer and little demand (e.g., Amazonas) should restrict the market to the state area, but issue unprotected OGF certificates to be traded with states with high demand for CRAs (e.g., Mato Grosso). Such strategy could make the CRA market a more attractive option for landholders with use rights over unprotected OGFs to negotiate their surpluses. Additional programs for payment for ecosystem services might also be a promising alternative to compensate OGF conservation that can easily be implemented using the CRA trade platform (Soares-Filho et al. 2016).

As Freitas et al. (2017a), we found that private properties are critical for the conservation of OGF carbon stocks in the Brazilian Amazon, stressing the importance of BFC compliance. Despite the expressive contribution of RFs for the increase in BFC compliance, we found that current (actual) carbon stocks in RFs are nearly negligible when compared to carbon storage of OGFs and add little change to the regulatory setups carbon balance. This is mostly due to the young age of RFs (Supplementary Figure III-2a). Important to highlight, the lack of a longer time series of RFs age information likely led to an underestimation of our carbon sequestration estimates in RFs. However, if we consider the attainable carbon storage by RFs, their future contribution would be meaningful, leading to a threefold increase in the protection of carbon stocks in setup 8, baseline 2, for example (Supplementary Table III-7).

Our results are consistent with previous studies, which found an imbalance between forest certificates offer and demand (i.e., deficits), possibly leading to an oversupply of forest certificates under a biome-wide market setup. Still, these assessments used different methods, datasets and assumptions, making direct comparisons tricky (Supplementary Table III-8). First, previous estimates of the BFC balance either excluded RF (Freitas et al. 2017b; Soares-Filho et al. 2016) or did not discriminate it from OGFs (Nunes et al. 2016; Soares-Filho 2013), whereas we make this distinction explicit and highlight the implications of a differentiation between old-growth and secondary vegetation. Second, we covered less area than most analysis, as only properties registered within the CAR were included. Other studies circumvented this limitation simulating properties for the remainder of the non-registered area (Freitas et al. 2017b; Micol et al. 2013) or using other spatial units as proxies for properties (Martini et al. 2015; Soares-Filho et al. 2014). This explains why our estimated CRA offer and forest deficits are smaller than calculated by other studies (Supplementary Table III-9).

We found a smaller certificate "offer-demand ratio" than previous research (Supplementary Table III-9). This is likely because we assessed rural settlement properties' individually (i.e., they are included if registered within the CAR), while other studies calculated the forest balance for whole settlements as units. Depending on the premise of the study (i.e., if settlements can supply certificates for the CRA market), rural settlements add substantial area to the pool of protected forests apt for compensation, and very little to deficits (Brito 2017). In addition, the full inclusion of rural settlements by other studies partially explains the higher proportion of protected forests apt for compensation than unprotected surplus in previous reports in comparison to this study (Micol et al. 2013; Nunes et al. 2016; Soares-Filho et al. 2014) (Supplementary Table III-9).

There are high uncertainties regarding the lawfulness of CAR entries located in public non-designated areas, which later may not be recognized as private (e.g., might be designated as indigenous or protected areas). This means that we might have overestimated unprotected surpluses from properties registered in public lands, especially properties located in remote, forested areas. Other studies dealt with this limitation either excluding the full surplus of states with large tracts of non-designated public lands (Soares-Filho et al. 2014) or blocking the simulation of properties in areas with more than 95% of forest cover (Freitas et al. 2017b). Different to these analyses we decided to include all the CAR entries to support the discussion based on the most accurate property level information available but highlight the respective uncertainties inherent to the data.

## 4.1. Challenges for implementation

Recent research supports the hypothesis that landholders might use ongoing forest-recovery to solve their deficits under certain conditions (Pacheco et al. 2017). In the Brazilian Amazon, regrowth usually takes place on marginal lands, where expected returns are sufficient to drive deforestation, but actual profits do not compensate production costs (Costa 2016). These areas may present a high aptitude for natural (passive) regeneration, which needs less investment compared to active restoration strategies required to recover very degraded ecosystems. In fact, a recent policy brief assessed that 60% of forest deficits in the Amazon have high potential for recovery through natural regeneration (MMA 2017c). If provided with the necessary incentives (e.g., facilitated access to credit lines, participation in complementary PES schemes), a large share of RF could be preserved at low costs. In this regard, synergies between agriculture and environmental policies are expected and could be beneficial. Synergistically with the BFC enforcement, the Low Carbon Agriculture Program created a credit line to support LR and PPA recovery in rural properties, aligned with the objectives of the PLANAVEG and the Brazilian NDC. However, in 2017 only 1% (US$4M) of the available resources was granted for this purpose. Therefore, the protection of regrowing vegetation depends upon the opportunity costs of lands where regrowth is taking place compared to the perceived noncompliance costs (i.e., credit restriction, fines). If the opportunity costs exceed the perceived compliance costs, it is not reasonable to expect that regrowing areas will be spared, as discussed by Aguiar et al. (2016).

The BFC still lacks mechanisms to support the protection of recovering forests (Garcia et al. 2016; Metzger et al. 2010) making state legislations key instruments to enable restoration targets. First, it is necessary to provide a comprehensive definition of second-growth forests to be protected, supported by forestry and ecological parameters. Such parameters should ideally support monitoring systems using remote sensing products to allow law enforcement and avoid conversion of second-growth forests. The protected status of second-growth forests should also be sensitive to social actors' practices distinguishing fallowing from land abandonment, to avoid the imposition of complicated licensing schemes on smallholders practicing swidden agriculture, that depend on cyclic forest regrowth, and avoid negative social impacts (Aguiar et al. 2016). A revision of the state-level PRAs showed that few legislations already place second-growth forests as native vegetation, differentiating them from degraded lands, but only the state of Pará details which second-growth forests should be protected and clearly establishes that issuing CRAs from RFs is allowed (Supplementary

Table III-3). Therefore, state laws should also regulate the use of RFs on LRs and compensation schemes to avoid competition with OGFs in the CRA market.

Despite the potential availability of apt RFs, we argue that a massive inflow of certificates issued from RFs to a CRA market is unlikely under any of the analyzed regulatory setups. The increasing scarcity of land for agricultural expansion, combined with the legal uncertainty of RFs protection and a saturated CRA market (e.g., setups 3-4), may direct landholders to use current secondary forests to expand productive area. With less land available, it is doubtful that farmers will source unprotected OGF, let alone their RF surplus for trade at an oversupplied market offering small prices for certificates. This suggests that inequality reduction and income redistribution may be a limited co-benefit of a CRA market in states where smallholder forest certificates offer is dominated by RFs, such as TO and MA (Supplementary Figure III-3). Freitas et al. (2017b) suggest that one alternative to promote social welfare through the BFC implementation would be to restrict the market to smallholdings. This could be one example of a policy tailored to target smallholders, less responsive to past policies directed to reduce deforestation (Godar et al. 2014; Richards and VanWey 2016). Other factors limiting farmers adherence to the certificate market could be land tenure insecurity (Brito 2017) or the lack of knowledge about the system (Rasmussen et al. 2016).

## 4.2. Study limitations and recommendations for future research

The BFC balance and market regulatory setups analyzed by this study are intended to be illustrative of the potential RFs offer to increase law compliance and carbon storage. However, there are no guarantees that RFs mapped by TerraClass are suitable for supporting a successful forest restoration plan. Mapping RFs is a central challenge (Caviglia-Harris et al. 2014), and despite being a big step forward for vegetation regrowth monitoring, the TerraClass product has limitations. TerraClass does not rely on information about successional status or land use history to detect forest cover expansion. For example, Nunes et al. (2016) found that large areas of RFs mapped in 2010 had been deforested only two years before, which is incompatible with the advanced stage of recovery TerraClass claims to detect (Almeida et al. 2016). Research relying on long-term time-series of satellite data could offer more consistency and allows to track indicators of RFs successional stage, such as time since abandonment (Carreiras et al. 2014; Müller et al. 2016b).

We calculated LR forest deficits based on an accumulated clear-cut deforestation map, which does not include forest losses due to degradation processes that may also require restoration.

Between 2007 and 2013 OGF degradation from fires or logging affected nearly twice the (clear-cut) deforested area in the Brazilian Amazon (INPE 2014a). This makes degradation processes a non-negligible source of carbon as well as a potential sink under an efficient forest governance scenario (Aguiar et al. 2016). However, as we lack up-to-date information on forest degradation, as well as regrowth dynamics following forest degradation, this remains an information gap to be addressed by future studies.

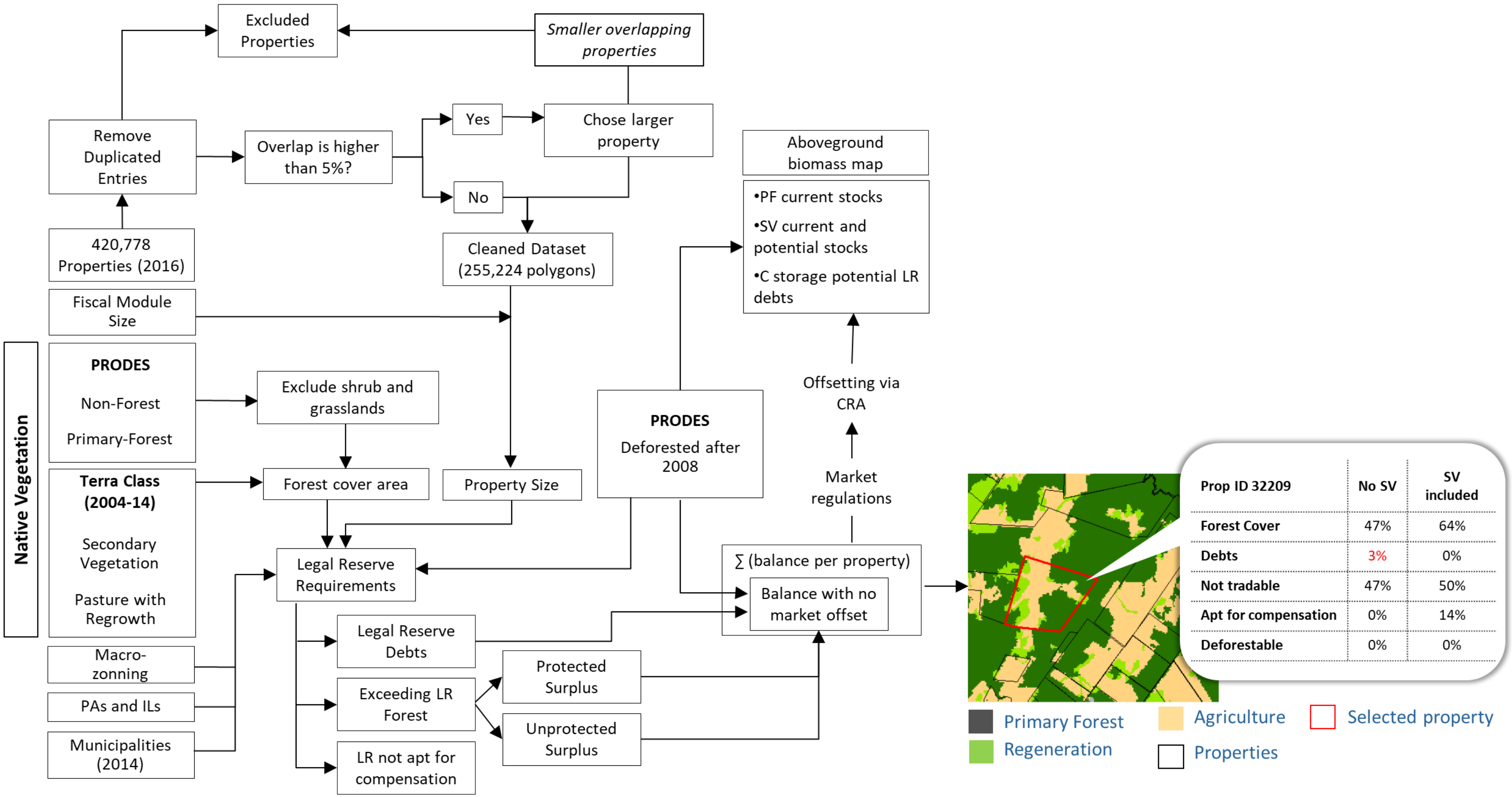
Forthcoming improvements of the CAR dataset may impact future assessments of the BFC balance. The CAR, as made public by the Brazilian Government, does not include property ownership information. Such data is important to accurately assess the extent of forest deficits and eligible area for issuing CRAs as landowners may purchase multiple properties to offset deficits from landholdings with LR area below the required cap. We stress that a thorough policy evaluation will benefit from property ownership data transparency. This holds true not only for BFC analyzes but also for related policies (e.g., Soy Moratorium) (Gollnow et al. 2018). Additionally, the CAR validation should solve current data inconsistencies (i.e., overlapping properties, duplications or false geometries) and allow the inclusion of all properties submitted to the SICAR system (SICAR 2017) in BFC assessments.

In this study we mapped ongoing recovery potential contribution to law compliance, OGFs and RFs (and associated carbon stocks) conservation. However, despite the enthusiasm on the potential offered by passive (natural) forest recovery (Crouzeilles et al. 2017b) an effective cross-sector implementation of the BFC (with other policies covering forest recovery and climate change) would strongly benefit from an investigation of synergic combinations of OGFs and RFs conservation and forest restoration based on indicators at multiple scales. For example, several recovering forest-patches may have been abandoned due to an advanced stage of soil degradation, and, therefore, could require more intervention than passive restoration offers. On that matter, Arroyo-Rodriguez et al. (2017) stressed that forest succession is driven by numerous factors interacting across local, landscape and regional levels. Future strategies for large-scale forest recovery should consider the synergies between land use and intensity history, forest connectivity, law compliance, carbon storage potential, topography and ecoregional conservation status.

## 5. Conclusions

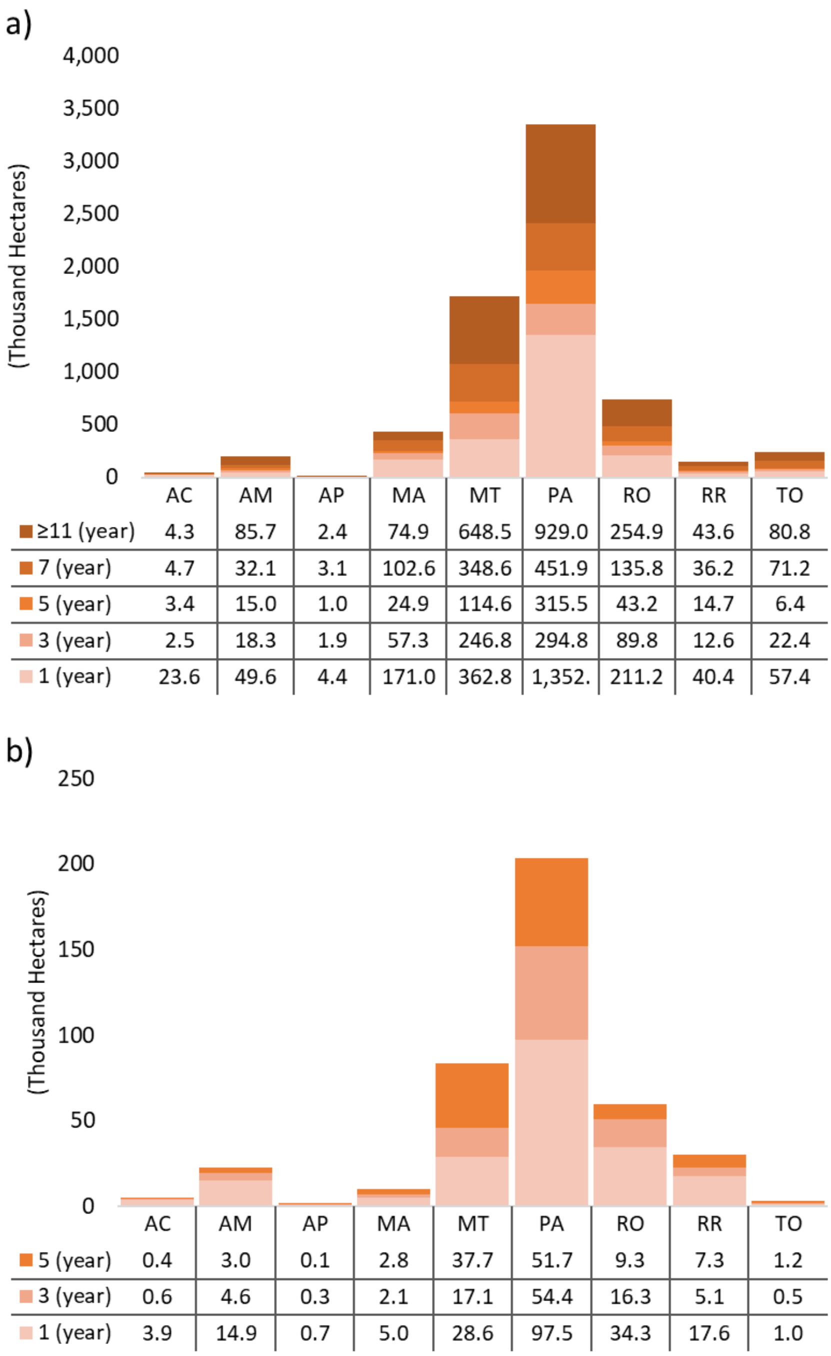
Geospatial information on property level have been enabling rigorous examinations of land use policies, including the BFC. Building up on previous detailed assessments our analysis shed light on an important aspect of the BFC law implementation: the contribution of ongoing forest recovery to the offer of forest certificates and forest deficits offsets across properties in the Brazilian Amazon. Our findings suggest that RFs may play an important role on deficit offsetting and forest certificates supply. Most important, policy outcomes differed drastically among the regulatory setups of the forest certificate trade mechanism here analyzed. Our results call attention for the need to explicitly include regrowing forests in BFC balance assessments, to support the design of state specific policies to maximize synergies between BFC law compliance, conservation additionality and forest recovery in degraded ecosystems of the Brazilian Amazon.

## 6. Supplementary material




| Prop ID 32209 | No SV | SV included |
|---|---|---|
| Forest Cover | 47% | 64% |
| Debts | 3% | 0% |
| Not tradable | 47% | 50% |
| Apt for compensation | 0% | 14% |
| Deforestable | 0% | 0% |

**Supplementary Figure III-1. Methodological Workflow.**



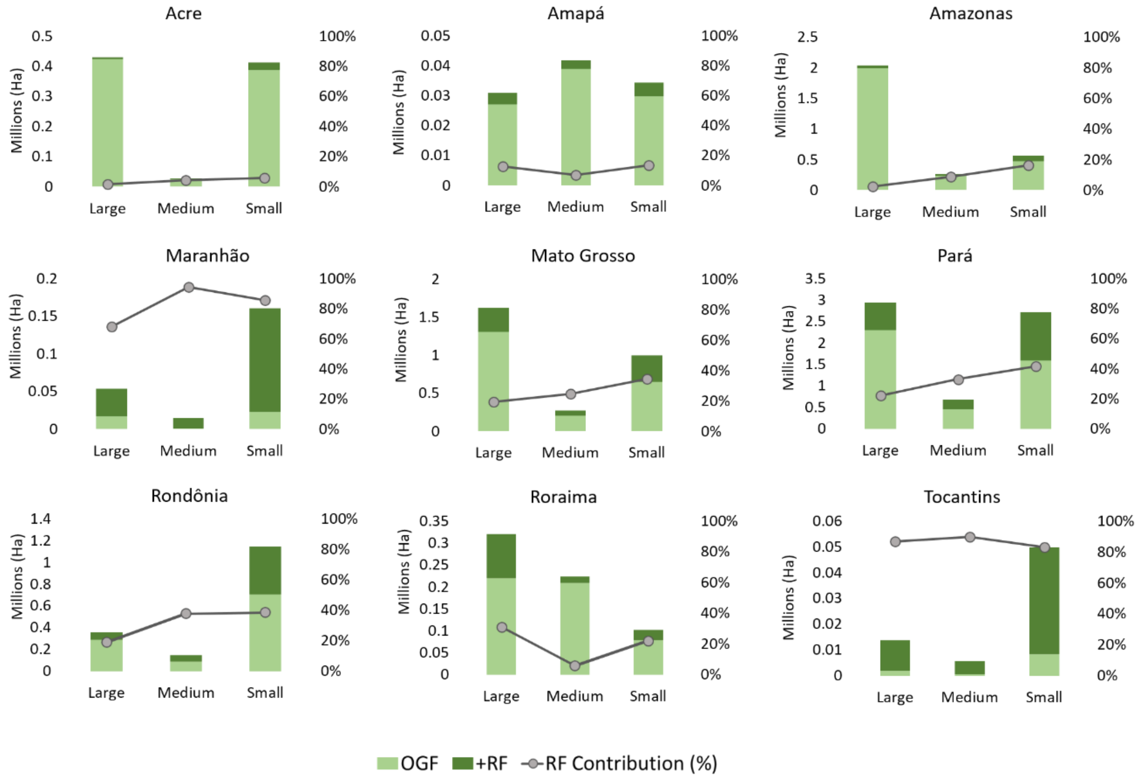

a)

| | AC | AM | AP | MA | MT | PA | RO | RR | TO |
|---|---|---|---|---|---|---|---|---|---|
| ≥11 (year) | 4.3 | 85.7 | 2.4 | 74.9 | 648.5 | 929.0 | 254.9 | 43.6 | 80.8 |
| 7 (year) | 4.7 | 32.1 | 3.1 | 102.6 | 348.6 | 451.9 | 135.8 | 36.2 | 71.2 |
| 5 (year) | 3.4 | 15.0 | 1.0 | 24.9 | 114.6 | 315.5 | 43.2 | 14.7 | 6.4 |
| 3 (year) | 2.5 | 18.3 | 1.9 | 57.3 | 246.8 | 294.8 | 89.8 | 12.6 | 22.4 |
| 1 (year) | 23.6 | 49.6 | 4.4 | 171.0 | 362.8 | 1,352. | 211.2 | 40.4 | 57.4 |

b)

| | AC | AM | AP | MA | MT | PA | RO | RR | TO |
|---|---|---|---|---|---|---|---|---|---|
| 5 (year) | 0.4 | 3.0 | 0.1 | 2.8 | 37.7 | 51.7 | 9.3 | 7.3 | 1.2 |
| 3 (year) | 0.6 | 4.6 | 0.3 | 2.1 | 17.1 | 54.4 | 16.3 | 5.1 | 0.5 |
| 1 (year) | 3.9 | 14.9 | 0.7 | 5.0 | 28.6 | 97.5 | 34.3 | 17.6 | 1.0 |

**Supplementary Figure III-2. a) Regrowing forests age in selected private properties in areas deforested prior to 2008, b) Regrowing forests age in selected private properties in areas deforested in or after 2008.** Source: PRODES and TerraClass programs (INPE 2014b; INPE 2014c)




**Supplementary Figure III-3. Forests stocks apt for compensation categorized by property size.** Large (> 15 FM), Medium (4>FM<15), Small (<4FM).

**Supplementary Table III-1. Description of concepts introduced by the BFC that are relevant to this study.**

| Concept | Acronym Used | Definition |
|---|---|---|
| *Legal reserve* | LR | Area located inside a rural property for native vegetation protection, ecological processes recovery and biodiversity conservation. It may include PPA areas for the purpose of law compliance increase. The extent set aside as LR is a proportion of the landholding area, which varies according to the location (biome), the vegetation type, property size, municipality protection level and rural consolidation status. |
| *Permanent Protection Area* | PPA | Riparian Zones of Rivers and Hilltops. May be covered or not by native vegetation, serving the purposes of resources (water landscape, terrain stability, biodiversity and soil) conservation and human population well-being. Can be added to the LR for compliance achievement, if it does not mean freeing land elsewhere in the property for deforestation. |
| *Plano de Regularização Ambiental* | PRADA | Compliance plan to be submitted by the indebted farmer to the pertinent state environmental institution detailing the pathways for regularization in accordance with the state PRA. |
| *Programa de Regularização Ambiental* | PRA | Legislation specifying the implementation of the forest code (BFC) on State level |
| *Environmental Reserve Quotas* | CRA | Vegetation cover certificates, which can be traded for off-site LR deficits compensation. In the case of private properties with LR surplus, CRAs can be created from areas with existing or regenerating vegetation cover (unless regeneration is unlikely to succeed, as verified by the Environmental Agency in charge). 1 CRA equals to 1 hectare of native vegetation. |
| *Legal reserve deficit and illegal deforestation* | - | Forest shortfall on a property according to the Law 12651/2012, either due to insufficient LR and/or due to illegal deforestation (taking place after 2008). Compliance with the law requires compensation on-site via restoration (mandatory for forest losses after 2008) or off-site (via CRA acquisition). |
| *Protected Forest - LR not apt for compensation* | - | LR in shortage or equaling the Law 12651/2012 requirements, hence not being apt for use in compensation schemes – except for small properties. |
| *Protected Forest – LR apt for compensation* | - | Share of forest in a property above the required by the regularization regime and below the required by the conservation regime of the BFC which cannot be converted for alternative uses but is apt for trade in a CRA market. |
| *Unprotected Forest Surplus* | - | Share of forest in a property above the required by the regularization and conservation regimes of the BFC which may be converted to alternative uses (if licensed by the government) but is also apt to be traded as forest certificates (CRA) in a BFC deficit compensation market. |

**Supplementary Table III-2. Conservation and regularization requirements used in this study in accordance with the BFC** (Law 12.651/2012).

| Criteria | **LR - Regularization Regime** | | **LR - Conservation Regime** | |
|---|---|---|---|---|
| | **(1)** Property Size ≤ 4 rural fiscal modules | **(2)** Property Size > 4 rural fiscal modules | **(3)** Property Size ≤ 4 rural fiscal modules | **(4)** Property Size > 4 rural fiscal modules |
| **(a)** No special conditions | LR *equals the amount of forest cover before 2008*. (Art. 67 of the BFC - Law 12.651/2012). | If LR is less than the required, the *debts must be restored or compensated to an amount equivalent to 80% of the properties as LR*. | All forest cover covering less or 80% of the property is eligible for compensation (*compensation surplus*). In case there is more than 80% of forest cover this amount is eligible for compensation or may be deforested, if properly licensed (*deforestable or compensation surplus*). | In this case, only the forest cover above 80% is eligible for compensation or may be deforested, if properly licensed (*deforestable or compensation surplus*). |
| **(b)** Protection Level | *The same as (a.1)* | Properties located in a municipality with more than 50% of its area included in protected areas *should restore only up to 50% of the properties as LR*. | *The same as (a.3)* | In this case, the forest below the 50% mandatory regularization LR is not eligible for compensation schemes. However, in case there is LR forest covering between 50-80% of the property, this amount is eligible for compensation schemes, but not for deforestation (*compensation surplus*). Finally, in case there is forest cover above 80% of the property is eligible for compensation or may be deforested, if properly licensed (*deforestable or compensation surplus*). |
| **(c)** Consolidation Level | *The same as (a.1)* | Properties located in a consolidated area zone according to the state's ecological economic zoning *should restore only up to 50% of the properties as LR*. | *The same as (a.3)* | *The same as (b.4)* |
| **(d)** Deforestation date | Forests illegally deforested after 2008 are required to be restored, regardless of the amount of LR in the property. | | In case of deforestation after 2008 being present at the property, then any forest surplus is not eligible for compensation until restoration takes place. | |

**Supplementary Table III-3. Main provisions in *Programas Estaduais de Regularização Ambiental* (PRA) to the eligibility of regrowing forests to LR deficit compensation via Environmental Reserve Quotas (CRA) acquisition.**

| State<br>Reference to the law | Relevant provisions<br>(free translation from Portuguese) | Interpretation |
|---|---|---|
| **Acre**<br><br>Lei nº 3.349 de 18 de dezembro de 2017 | Art. 8º. The LR regularization will be made through "*recomposição*" (i.e., reestablishment of a vegetation cover using a mix of native and exotic species with commercial value), compensation or a combination of both.<br>§ 5º - The LR compensation (...) will be allowed via the following mechanisms<br>I - Acquisition of Environmental Reserve Quotas i.e., CRA<br>II - Lease of LRs or areas under regime of "*servidão ambiental*" i.e., environmental easement located in third party properties<br>III - Donation of areas located in conservation units pending regularization to the State.<br>IV - Registration of an equivalent area, additional to the LR, covered by established native vegetation, under recovery, in a property or landholding of the same titularity or owned by a third party, if located in the same biome.<br>§ 6º - Requirements defining the aptitude of an area for compensation is the equivalence of the area in hectares, location in the same biome and, if in a different state, the area should be listed as priority for conservation by state or federal governments. | Art. 8 defines that compensation is one of the means through which LR regularization may be achieved. The article describes three types of compensation, being the acquisition of CRAs one of them. However, the state law makes no further restrictions to the federal definition of CRA (see article 46 Items I-II of Native Vegetation Protection Law 12651/2012). Therefore, CRAs issued from RFs should be eligible for LR deficit compensation in Acre or subject to future regulations. |
| **Amazonas**<br><br>Lei nº. 4.406, de 28 de dezembro de 2016 | Art. 11º § 1º. Item IV - The CRA is an instrument of the PRA<br>Art. 30º. The landholder may offset LR deficits via Items<br>I - "*Recomposição*" i.e., reestablishment of a vegetation cover using a mix of native and exotic species with commercial value<br>II - Natural regeneration,<br>III - Compensation<br>(…) § 9º - The compensation mentioned in item III of Art 30 may take place in properties registered at the CAR in Amazonas state, in the Amazon Biome, with an equivalent area to the property LR deficit, using the following mechanisms:<br>I – Donation of areas located in conservation units pending regularization to the State.<br>II – Registration of an equivalent area, additional to the LR, covered by established native vegetation in stage of successional climax, in a property or landholding of the same titularity or owned by a third party<br>Art. 31º. The organization responsible for formulating the state-level environmental policy will define the areas in the state of Amazonas with priority for the compensation of LR deficits from non-compliant landholdings from other states as described by § 6º and § 7º of Art 66 of the Native Vegetation Protection Law 12651/2012. | The state legislation does not allow the use of CRAs for the compensation of deficits within the borders of Amazonas state. Instead, there are two other compensation mechanisms permitted (see Art 30 § 9, item I-II). However, the legislation foresees that CRAs may be issued from eligible forests located in properties of Amazonas state overlapping priority areas for conservation. Such CRAs may be sold to non-compliant farms in other states aiming to regularize their LR deficits. The state law makes no further restrictions to the federal definition of CRA (see article 46 Items I-II of Native Vegetation Protection Law 12651/2012). Therefore, there are no legal restrictions to issuing CRAs from RFs to be sold for offsetting deficits outside the state of Amazonas. |

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|---|---|---|
| **Amapá** | The state has not published the PRA | |
| **Maranhão**<br><br>Lei nº 10.276 de 07 de julho de 2015<br><br>Portaria SEMA nº 64 de 04 de agosto de 2014 | **Lei nº 10.276 de 07 de julho de 2015**<br>Art. 3º. The following are mechanisms for environmental regularization of properties and rural activities:<br>II – the TC (i.e., term of adjustment);<br>Art. 5º, Item II – Natural vegetation area reminiscent: includes areas covered by native vegetation, intact or under regeneration;<br>Art 7º. The TC has the goal of establishing the conditions and deadlines for the fulfillment of the legal requirements for environmental regularization;<br>§ 2º. when formalizing the TC, in case the recovery of PPAs or LR are necessary, the interested party should present:<br>I – Plan for the recovery of degraded areas or adhering to the recovery practices approved by the State Environmental Council Manual - CONSEMA<br>**Portaria SEMA nº 64 de 04 de agosto de 2014**<br>Describes the technical and administrative procedures for issuing and controlling the CRAs<br>Art. 5º. Each CRA corresponds to 1 hectare:<br>I – of area with primary or secondary vegetation at any stage of recovery<br>II – Areas actively recovered with native species<br>Art. 8 º, § 1º, the CRA will only be eligible for use to compensate LR in property located in the same biome where it is located, preferably in the state of Maranhão, or in another state, if the respective responsible environmental institution accepts the use of CRAs issued in Maranhão.<br>§ 2º, the CRA will only be allowed as a mean for compensation if the terms of art. 66 of the law 12.6551/ 2012 are respected. | The Law n. 10.276 institutes the PRA in Maranhão. However, the text is very vague and does not mention the possibility of off-site compensation of LR deficits nor the CRA as a mechanism for regularization of LR deficits. The possibility of issuing CRAs from eligible forests for trade in Maranhão or in other states is also not mentioned. However, a legal document from 2014 (Portaria SEMA nº 64 de 04 de agosto de 2014) regulates the issuing of CRAs in Maranhão, and stresses that both primary and secondary may be used to issue CRAs. The document also highlights that CRAs may be used to compensate deficits in Maranhão (or elsewhere depending on other states regulations) and refers to the federal legislation for further guidance. It is not clear if the 2014 *Portaria* is still valid. New regulations could set the guidelines for LR compensation in the future. |
| **Mato Grosso**<br><br>Lei Complementar nº. 592, de 26 de maio de 2017 | Art. 2º. For the purposes of this decree, the following terms are defined:<br>V – Altered area: Area that, after natural impact or anthropic intervention, still maintains the capacity to naturally regenerate, to a condition that might be different than the original;<br>VI – Degraded area: Area that is altered due to anthropic intervention with no capacity of natural regeneration;<br>Item XIV – Environmental Regularization: activities implemented in the rural property aiming to conform with the environmental legislation, and, mostly, to maintain and recovery the PPAs, LRs and restricted use areas, and the compensation of LR deficits, when appropriate;<br>Art. 23º, § 3º the landowner or landholder of a property with conserved LR, registered at the CAR (...), exceeding the minimum conservation requirements will be allowed to use the surplus area for (...) issuing CRAs and for compensation (...);<br>Art. 25º. The Adjustment Term or similar instruments for environmental regularization of rural properties dealing with the PPAs, LRs and restricted use areas created under the rule of past legislation should be revised to conform to the Native Vegetation Protection Law 12651/2012. | The compensation of LR deficits is allowed in Mato Grosso, so as the use and issuing of CRAs. Any further guidance should refer to the Native Vegetation Protection Law 12651/2012. Therefore, there are no legal restrictions to issuing CRAs from RFs to be sold for offsetting deficits outside the state of Mato Grosso. |
| **Pará** | Art. 1º, § 2º the following are instruments of the PRA: IV – the CRAs, when appropriate<br>Art. 2º. For the purposes of this decree, the following terms are defined:<br>II - Altered area: Area that, after an impact maintains its capacity of natural regeneration;<br>III – Degraded area: altered area, due to anthropic impact, with no capacity for natural regeneration; | The compensation of LR deficits is allowed in Pará, so as the use and issuing of CRAs. The state legislation specifically mentions that CRAs may be issued from recovering forest |

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| Decreto nº. 1379 de 03 de setembro de 2015 | IV – native vegetation reminiscent: all fragments of native vegetation existing in the property, both primary and secondary; the secondary requires a classification of the regeneration stage, following specific regulation.<br>VI – CRA: A landowner-binded title representing the area with standing native vegetation cover or under recovery, as described by the Native Vegetation Protection Law 12651/2012.<br>Art. 31º. The landowner/ landholder of a rural property with native vegetation falling short on the minimum required to constitute the LR shall adopt one or more of the following measures:<br>I – Reestablish native vegetation via active restoration;<br>II – Conduct a natural regeneration process;<br>III – compensate the LR deficit.<br>Art. 39º. The LR deficit regularization may occur through compensation via:<br>(...) III – acquisition of CRAs;<br>Art. 40º. The areas to be used for LR compensation shall:<br>I – be equivalent in area extent to the LR to be compensated;<br>II – be located in the same biome as the LR to be compensated;<br>III – if outside the state of Pará, they shall be in areas defined as priority by the Federal or State power. (…) depending on the presentation of technical or economic proof of the inability to compensate LR déficits within the borders of the state of Pará.<br>Art. 62º. The landowner of a rural property a conserved LR beyond the minimum conservation requirements will be eligible to issue CRAs representing the area covered by native vegetation, primary or under an intermediate-advanced stage of recovery according to the specific regulation and classification. | areas (in intermediate to advanced recovery stage). Such CRAs from recovering forests may also be used for compensation. The law also refers to specific legislation that should be used as a guideline to classify secondary forests successional stages. |
| **Rondônia**<br><br>Decreto nº 20.627 de 08 de março de 2016 | Art. 1º, the following are instruments of the PRA: V – the CRAs<br>Art. 2º. For the purposes of this decree, the following terms are defined:<br>V – Degraded area: altered area, due to anthropic impact, with no capacity for natural regeneration;<br>VI - Altered area: Area that, after an impact maintains its capacity of natural regeneration;<br>Art. 31º. The landowner/ landholder of a rural property with native vegetation falling short on the minimum required to constitute the LR shall adopt one or more of the following measures:<br>I – Reestablish native vegetation via active restoration;<br>II – Conduct a natural regeneration process;<br>III – compensate the LR deficit.<br>Art. 39º. The LR deficit regularization may occur through compensation via:<br>(...) III – acquisition of CRAs;<br>Art. 40º. The areas to be used for LR compensation shall:<br>I – be equivalent in area extent to the LR to be compensated;<br>II – be in the same biome as the LR to be compensated as defined by the Instituto Brasileiro de Geografia e Estatística - IBGE;<br>III – if outside the state of Pará, they shall be in areas defined as priority by the Federal or State power.<br>Art. 43º. The landowner of a rural property a conserved LR beyond the minimum conservation requirements will be eligible to (...) and issue CRAs.<br>Art. 54º. The issuing of CRAs will follow the regulations by the Executive Federal power. | The compensation of LR deficits is allowed in Rondônia, so as the use and issuing of CRAs. The state legislation cites the federal law as reference for further guidance. Therefore, there are no legal restrictions to issuing CRAs from RFs to be sold for offsetting deficits outside the state of Rondônia. |

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|---|---|---|
| **Roraima** | The state has not published the PRA | |
| **Tocantins**<br><br>Lei nº 2713 de 09 de maio de 2013 | Art. 3º. the following are mechanisms for environmental regularization of properties and rural activities:<br>II – the TC (i.e., term of adjustment);<br>Art 5º. Item I, c), 1. Areas with natural vegetation reminiscent – include areas covered by native vegetation, intact or under regeneration;<br>Art 7º. The TC has the goal of establishing the conditions and deadlines for the fulfillment of the legal requirements for environmental regularization;<br>§ 1º - O TC deve estipular obrigações para o atendimento das exigências destinadas à regularização tempestiva da Reserva Legal<br>§ 2º - when formalizing the TC, in case the recovery of PPAs or LR are necessary, the interested party should present:<br>I – Plan for the recovery of degraded areas or adhering to the recovery practices approved by the State of Tocantins Environmental Council - COEMA-TO; | The Law nº 2713 de 09 de maio de 2013 institutes the PRA in Tocantins<br>. However, the text is very vague and does not mention the possibility of off-site compensation of LR deficits nor the CRA as a mechanism for regularization of LR deficits. The possibility of issuing CRAs from eligible forests for trade in Tocantins<br>or in other states is also not mentioned. New regulations could set the guidelines for LR compensation in the future. |

**Supplementary Table III-4. Datasets description.**

| Information | Definition |
|---|---|
| **Municipalities**[12] | Administrative sub-divisions of the Brazilian Federal States. The Brazilian Legal Amazon is formed by 775 municipalities. |
| **Fiscal Modules**[13] | A measurement unit, in hectares, defined by INCRA (*Instituto Nacional de Colonização e Reforma Agrária*) as the minimum average size of economically viable properties within a municipality. It reflects the geographical location, predominant rural activity, income and familial unit characteristics. It ranges from 5 to 110 hectares across the country. The fiscal module is the official criterion used to classify properties by size (1-4 fiscal modules = small properties; 4-15 fiscal modules = medium properties; > 15 modules = large properties). |
| **Restricted Land Use Categories** | *Here Protected Areas include Protected Areas, Indigenous Lands and Military Areas. We divided the BLA territory in two broad categories: protected and not-protected to calculate the municipality's "level of protection" (i.e., share of the municipality included in protected areas). If a municipality is > 50% protected than medium and large landholders have the right to flexible conditions to LR regularization and LR conservation (Art. 12 § 4o of the BFC - Law 12.651/2012).*<br><br>**Protected Areas**[14]: The network of federal, state and municipal protected areas divided in 12 categories, differing in land use restriction level and conservation purposes. All categories were included, except for the APA (Environmental Protection Areas) due to their relaxed restrictions.<br><br>**Indigenous Lands**[15]: Territories traditionally occupied or designated to Indigenous groups in Brazil. Due to their permanent use unavailability by other groups in society they can be considered as protected areas. |
| **Consolidated Uses** | The Art. 13 § 8o of the BFC (Law 12.651/2012) establishes that a property inserted in an area defined by the state level Economic Ecological Zoning (ZEE acronym in Portuguese) as consolidated may have its LR requirement reduced to 50%, under the regularization regime. For this study we identified areas zoned as consolidated as mapped by the Legal Amazon Economic Ecological Macro Level Zoning[16] (established by Federal Decree nº 7.378/2010). |
| **Deforestation**[17] | The PRODES assessment monitors clear-cut deforestation in the Brazilian Legal Amazon since 1988, providing annual estimates. Since 2000, PRODES annually provides spatially explicit estimates (i.e., a cumulative deforestation raster dataset discretizing the year of the clearing event). For our study we used deforestation information up to 2014. |
| **Primary Forests**[18] | We used the old-growth forest mask provided by PRODES (*See above*), corresponding to standing pristine forest in 2014. |

[12] IBGE – *Instituto Brasileiro de Geografia e Estatística.* https://mapas.ibge.gov.br/bases-e-referenciais/bases-cartograficas/malhas-digitais.html
[13] INCRA - *Instituto Nacional de Colonização e Reforma Agrária*. http://www.incra.gov.br/tabela-modulo-fiscal
[14] MMA (*Ministério do Meio Ambiente*). http://mapas.mma.gov.br/i3geo/datadownload.htm
[15] FUNAI - Fundação Nacional do Índio.http://www.funai.gov.br/
[16] MMA (*Ministério do Meio Ambiente*). http://mapas.mma.gov.br/i3geo/datadownload.htm
[17] PRODES. INPE – Instituto Nacional de Pesquisas Espaciais.
[18] PRODES. INPE – Instituto Nacional de Pesquisas Espaciais.

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|---|---|
| **Secondary Vegetation (here used as synonym for regrowing forests)**[19] | Terra Class provides *detailed maps of land use land cover of the deforested areas of the Brazilian Legal Amazon. Land uses were mapped with Landsat-5/TM images using techniques, such as linear spectral mixture model, threshold slicing and visual interpretation, aided by temporal information extracted from NDVI MODIS time series (Almeida et al. 2016 p. 291).*<br><br>To create a map of secondary vegetation cover, we included two classes obtained from TerraClass, as they both were covered by the concept of regenerating vegetation defined by the BFC:<br><br>*Regeneration with pasture:*<br>*Areas that were clear-cut, later developed as pasture and are at the beginning of a regenerative process containing shrubs and early successional vegetation (Almeida et al. 2016 p. 293).*<br><br>*Secondary vegetation:*<br>*Areas that were clear-cut and are at an advanced stage of regeneration with trees and shrubs. Includes areas that were used for forestry (silviculture) or permanent agriculture with use of native or exotic species. (Almeida et al. 2016 p. 293).* |

**Supplementary Table III-5. Tested market regulatory setups (scenarios) description.**

| | |
|---|---|
| **Scenario 1** | Protected LR apt for CRA and Unprotected forest surplus + Restricted to state borders + Old-Growth Forest (OGF) only |
| **Scenario 2** | Protected LR apt for CRA and Unprotected forest surplus + Biome Restricted + Old-Growth Forest (OGF) only |
| **Scenario 3** | Protected LR apt for CRA and Unprotected forest surplus + Restricted to state borders + Old-Growth Forest (OGF) and regrowing forests (RF) |
| **Scenario 4** | Protected LR apt for CRA and Unprotected forest surplus + Biome Restricted + Old-Growth Forest (OGF) and regrowing forests (RF) |
| **Scenario 5** | Unprotected forest surplus + Restricted to state borders + Old-Growth Forest (OGF) only |
| **Scenario 6** | Unprotected forest surplus + Biome Restricted + Old-Growth Forest (OGF) only |
| **Scenario 7** | Unprotected forest surplus + Restricted to state borders + Old-Growth Forest (OGF) and regrowing forests (RF) |
| **Scenario 8** | Unprotected forest surplus + Biome Restricted + Old-Growth Forest (OGF) and regrowing forests (RF) |

[19] TERRACLASS INPE – Instituto Nacional de Pesquisas Espaciais. http://www.inpe.br/cra/projetos_pesquisas/dados_terraclass.php

**Supplementary Table III-6. a) Carbon dioxide ($PgCO_2e$) content in forests in selected private properties per category; b) Carbon dioxide ($PgCO_2e$) potential uptake (total) of deficit categories;** ($PgCO_2e = MtCO_2e*10^3$)

| | Class | PF | RF (Actual)* | RF (Potential)* | Total (PF+RF) ** |
|---|---|---|---|---|---|
| a) Forest | Protected forests (not apt for CRA) | 13.5 | 0.1 | 1.6 | 13.6 |
| | Protected forests (apt for CRA) | 5.3 | 0.1 | 2.0 | 5.4 |
| | ***Sub-Total (Protected)*** | ***18.8*** | ***0.2*** | ***3.6*** | ***19.0*** |
| | Unprotected forest surplus | 1.9 | 0.0*** | 0.3 | 1.9 |
| | Regenerating forests in post-2008 deforestation areas | - | 0.0*** | 0.2 | 0.0*** |
| | **Total** | **20.7** | **0.2** | **3.8** | **20.9** |
| b) Deficit | LR Deficits | - | - | 3.7 | - |
| | LR Remaining Deficits (calculated including PF and RF in stocks) | - | - | 2.2 | - |
| | Illegal deficits in post-2008 deforestation areas | - | - | 1.0 | - |

* RF(Actual)=Regrowing forests current carbon content and RF(Potential) = Regrowing forests potential carbon content; ** sums are between PF and RF actual values; *** Value < 0.1.

**Supplementary Table III-7. Carbon dioxide ($PgCO_2e$) balance of different market regulatory setups (1-8).** Two baselines were set as references for additionality evaluation. ($PgCO_2e = MtCO_2e*10^3$)

| | Scenarios 1-2 | | Scenarios 3-4 | | Scenarios 5-6 | | Scenarios 7-8 | |
|---|---|---|---|---|---|---|---|---|
| | State | Biome | State | Biome | State | Biome | State | Biome |
| **Carbon Balance** (Baseline 1, ɸ) | -1.5 | -1.9 | -1.6 | -1.9 | -0.8 | 0.0 | -0.8 | 0.0 |
| **Carbon Balance** (Baseline 1, ɨ) | -1.5 | -1.9 | -1.6 | -1.9 | -0.8 | 0.0 | -0.6 | 0.3 |
| **Carbon Balance** (Baseline 2, ɨ) | -0.3 | -0.9 | 3.0 | 2.7 | 2.8 | 3.0 | 4.7 | 5.0 |

**Baseline 1** - Balance = Total $CO_2$ equivalent stocks in unprotected forest surplus minus the $CO_2$ equivalent stocks in the demand for CRAs issued from unprotected forest surplus.
**Baseline 2** - Balance = Total $CO_2$ equivalent stocks in unprotected forest surplus minus the $CO_2$ equivalent stocks in the demand for CRAs issued from unprotected forest surplus plus all carbon sequestration from regrowing forests assured by the BFC enforcement (i.e., remaining deficits regeneration + RF forests protection)
**ɸ** Carbon content of current regrowing forests corresponds to current (2014) values; **ɨ** Carbon content of current regrowing forests corresponds to potential values of pristine forests (e.g., sequestration potential.)

**Supplementary Table III-8. Total (actual and potential) forest carbon content ($MtCO_2e$) from selected private properties per category and potential sequestration ($MtCO_2e$) from deficits restoration.** Results per federal state: AC=Acre; AM=Amazonas; AP=Amapá; MA=Maranhão; MT=Mato Grosso; PA=Pará; RO=Rondônia; RR=Roraima; TO=Tocantins.; PF=Primary Forest; RF-Actual=Regrowing forests current carbon content; RF-Potential= Regrowing forests potential carbon content.

| | Forest or Deficit Category | AC | AM | AP | MA | MT | PA | RO | RR | TO | Total (Sum of all states) |
|---|---|---|---|---|---|---|---|---|---|---|---|
| (No CRA market) | Protected forests, not apt for CRA (PF) | 920.2 | 2,649.6 | 79.7 | 133.2 | 4,287.5 | 4,202.7 | 722.9 | 450.8 | 61.7 | 13,508.4 |
| | Protected forests, not apt for CRA (RF-Actual) | 0.2 | 1.5 | 0.0 | 9.1 | 31.4 | 52.3 | 6.7 | 0.3 | 6.1 | 107.6 |
| | Protected forests, not apt for CRA (RF-Potential) | 4.6 | 16.4 | 0.8 | 156.2 | 404.0 | 844.6 | 93.0 | 5.1 | 79.3 | 1,603.8 |
| | Protected forests apt for CRA (PF) | 380.4 | 1,051.1 | 49.6 | 24.7 | 575.7 | 2,414.4 | 514.6 | 240.4 | 5.0 | 5,256.1 |
| | Protected forests, apt for CRA (RF-Actual) | 0.6 | 5.1 | 0.3 | 5.9 | 16.6 | 71.5 | 21.0 | 5.8 | 2.0 | 128.9 |
| | Protected forests, apt for CRA (RF-Potential) | 17.0 | 66.8 | 5.3 | 112.9 | 223.2 | 1,189.7 | 290.3 | 79.8 | 23.9 | 2,008.8 |
| | Unprotected forest surplus (PF) | 119.5 | 722.4 | 20.2 | 2.0 | 386.5 | 480.8 | 69.1 | 50.5 | 0.1 | 1,851.1 |
| | Unprotected forest surplus (RF-Actual) | 0.2 | 3.0 | 0.8 | 0.4 | 6.7 | 9.6 | 1.9 | 0.4 | 0.1 | 23.1 |
| | Unprotected forest surplus (RF-Potential) | 2.9 | 40.2 | 2.5 | 6.7 | 78.2 | 151.6 | 24.4 | 4.5 | 1.0 | 311.9 |
| Potential sequestration from full deficit restoration | LR Deficits, Potential Carbon | 88.7 | 33.3 | 1.1 | 350.3 | 1,334.5 | 1,515.5 | 221.9 | 8.0 | 190.5 | 3,743.8 |
| | LR Remaining Deficits (calculated including PF and RF in stocks), Potential Carbon | 84.6 | 16.9 | 0.5 | 199.7 | 947.4 | 720.5 | 134.6 | 3.9 | 112.5 | 2,229.9 |
| | Deficits from post-2008 deforestation, Potential Carbon | 37.4 | 43.7 | 1.9 | 21.4 | 147.4 | 571.9 | 96.6 | 45.8 | 4.4 | 955.1 |
| | Regenerating forests in post-2008 deforestation areas, Actual Carbon | 0.1 | 0.3 | 0.0 | 0.2 | 1.4 | 4.4 | 0.8 | 0.5 | 0.1 | 7.7 |
| | Regenerating forests in post-2008 deforestation areas, Potential Carbon | 3.0 | 13.8 | 0.7 | 6.3 | 35.0 | 133.3 | 30.9 | 17.4 | 1.3 | 241.7 |

**Supplementary Table III-9. Mean carbon ($MtCO_2e$) content in demanded area for compensation per category and potential Carbon content ($MtCO_2e$) of remaining deficits that could not be offset by compensation according to different regulatory setups.** Values in brackets represent the minimum and maximum attainable carbon content.

| | **LR Category\ Unit** | **AC** | **AM** | **AP** | **MA** | **MT** | **PA** | **RO** | **RR** | **TO** | **Total (S1) (Sum of all states)** | **Biome (S2)** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Scenarios S1 and S2** | Protected forests apt for CRA (PF) | 80.8 (-16.3, +4.4) | 38.9 (-18.5, +12.5) | 1.7 (-1.6, +1) | 24.7 (0, +0) | 575.7 (0, +0) | 1653.4 (-123.9, +25.1) | 218 (-10.4, +3.4) | 9.5 (-7.0, +2.8) | 5.0 (0, +0) | 2607.8 (-197.6, +32.9) | 4269.6 (-124.1, +39.8) |
| | Protected forests, apt for CRA (RF-Actual) | | | | | | | | | | | |
| | Protected forests, not apt for CRA (RF-Potential) | | | | | | | | | | | |
| | Unprotected forest surplus (PF) | 0.0 | 0.0 | 0.0 | 2.0 (0, +0) | 386.5 (0, +0) | 0.0 | 0.0 | 0.0 | 0.1 (0, +0) | 388.6 (0, +0) | 0.00 |
| | Unprotected forest surplus (RF-Actual) | | | | | | | | | | | |
| | Unprotected forest surplus (RF-Potential) | | | | | | | | | | | |
| | Potential sequestration from restoration of LR remaining deficits | 0.0 | 0.0 | 0.0 | 323.7 (-4.8, +2.2) | 465.7 (-26.3, +14.9) | 0.0 | 0.0 | 0.0 | 30.2 (-6.0, +5.5) | 819.6 (-37.1, +22.6) | 0.00 |

| | **LR Category\ Unit** | **AC** | **AM** | **AP** | **MA** | **MT** | **PA** | **RO** | **RR** | **TO** | **Total (S3) (Sum of all states)** | **Biome (S4)** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Scenarios S3 and S4** | Protected forests apt for CRA (PF) | 77.0 (-16.8, +3.9) | 22.7 (-12.6, +8.7) | 1.0 (-1.0, +0.3) | 24.7 (0, +0) | 575.7 (0, +0) | 814.6 (-117.3, +24.5) | 132.5 (-12.0, +3.0) | 4.6 (-4.6, +3.3) | 5.0 (0, +0) | 1657.9 (-174.3, +43.7) | 2669.4 (-121.9, +42.2) |
| | Protected forests, apt for CRA (RF-Actual) | 0.0 | 0.0 | 0.0 | 5.9 (0, +0) | 16.6 (0, +0) | 0.0 | 0.0 | 0.0 | 2.0 (0, +0) | 24.5 (0, +0) | 0.00 |
| | Protected forests, not apt for CRA (RF-Potential) | 0.0 | 0.0 | 0.0 | 112.9 (0, +0) | 223.2 (0, +0) | 0.0 | 0.0 | 0.0 | 23.9 (0, +0) | 360.0 (0, +0) | 0.00 |
| | Unprotected forest surplus (PF) | 0.0 | 0.0 | 0.0 | 2.0 (0, +0) | 226.4 (-14.6, +8.9) | 0.0 | 0.0 | 0.0 | 0.1 (0, +0) | 228.5 (-14.6, +8.9) | 0.0 |
| | Unprotected forest surplus (RF-Actual) | 0.0 | 0.0 | 0.0 | 0.4 (0, +0) | 0.0 | 0.0 | 0.0 | 0.0 | 0.1 (0, +0) | 1.0 (0, +0) | 0.00 |
| | Unprotected forest surplus (RF-Potential) | 0.0 | 0.0 | 0.0 | 6.7 (0, +0) | 0.0 | 0.0 | 0.0 | 0.0 | 1 (0, +0) | 7.7 (0, +0) | 0.00 |
| | Potential sequestration from restoration of LR remaining deficits | 0.0 | 0.0 | 0.0 | 54.7 (-5.1, +1.8) | 0.0 | 0.0 | 0.0 | 0.0 | 4.1 (-2.9, +1.8) | 58.8 (-8.0, +3.6) | 0.00 |

| Scenarios S5 and S6 | LR Category\ Unit | AC | AM | AP | MA | MT | PA | RO | RR | TO | Total (S5) (Sum of all states) | Biome (S6) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Protected forests apt for CRA (PF) | | | | | | | | | | | |
| | Protected forests, apt for CRA (RF-Actual) | | | | | | | | | | | |
| | Protected forests, not apt for CRA (RF-Potential) | | | | | | | | | | | |
| | Unprotected forest surplus (PF) | 76.7 (-6.2, +4.3) | 38.6 (-14.8, +13.5) | 1.7 (-1.4, +0.3) | 2 (0, +0) | 386.5 (0, +0) | 480.8 (0, +0) | 69.1 (0, +0) | 9.9 (-2.3, +0.7) | 0.1 (0, +0) | 1065.4 (-48.5, +18.8) | 1851.1 (0, +0) |
| | Unprotected forest surplus (RF-Actual) | | | | | | | | | | | |
| | Unprotected forest surplus (RF-Potential) | | | | | | | | | | | |
| | Potential sequestration from restoration of LR remaining deficits | 0.0 | 0.0 | 0.0 | 348 (-4.4, +0.9) | 1006.5 (-22.1, +12.8) | 1086.6 (-12.7, +14.1) | 150.0 (-2.6, +2.1) | 0.0 | 34.6 (-6.2, +6.1) | 2625.0 (-48.0, + 36.1) | 2018.3 (-35, + 31.2) |

| Scenarios S7 and S8 | LR Category\ Unit | AC | AM | AP | MA | MT | PA | RO | RR | TO | Total (S7) (Sum of all states) | Biome (S8) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Protected forests apt for CRA (PF) | | | | | | | | | | | |
| | Protected forests, apt for CRA (RF-Actual) | | | | | | | | | | | |
| | Protected forests, not apt for CRA (RF-Potential) | | | | | | | | | | | |
| | Unprotected forest surplus (PF) | 73.0 (-6.5, +3.7) | 21.4 (-9.2, +12.4) | 1.1 (-1.0, +0.5) | 2.0 (0, +0) | 386.5 (0, +0) | 480.8 (0, +0) | 69.1 (0, +0) | 4.9 (-2.2, +0.6) | 0.1 (0, +0) | 1038.9 (-31.2, +17.3) | 1851.1 (0, +0) |
| | Unprotected forest surplus (RF-Actual) | 0.0 | 0.0 | 0.0 | 0.4 (0, +0) | 6.7 (0, +0) | 9.6 (-0.1, +0) | 1.9 (0, +0) | 0.0 | 0.1 (0, +0) | 18.8 (-0.1, +0.0) | 23.1 (0, +0) |
| | Unprotected forest surplus (RF-Potential) | 0.0 | 0.0 | 0.0 | 6.7 (0, +0) | 78.2 (0, +0) | 151.6 (-0.7, +0) | 24.4 (-0.2, +0) | 0.0 | 1 (0, +0) | 261.8 (-0.9, +0) | 311.9 (-7.6, +0) |
| | Potential sequestration from restoration of LR remaining deficits | 0.0 | 0.0 | 0.0 | 190.9 (-4.5, +1.2) | 549.3 (-19.7, +11.2) | 179.3 (-10.1, +6.8) | 37.6 (-2.6, +1.4) | 0.0 | 30.7 (-6.6, +4.5) | 987.8 (-43.5, +25.1) | 400 (-24.8, +16.5) |

**Supplementary Table III-10. Study Comparison.**

| *Study* | *Area* | *Study or selected scenario premises* | *Scale* | *Area (Mha)* | *Deficit** (Mha)* | *CRA eligible area (Mha)* | *Unprotected Surplus (Mha)* | *CRA offer/demand ration* | *Protected apt for CRA/Unprotected surplus Ration* |
|---|---|---|---|---|---|---|---|---|---|
| *Micol et al. (2013)* | MT | * Does not include SV in deficit and surplus calculation<br>* Includes Rural Settlements<br>* PRODES data | Property (203,000) CAR, LAU, Simulated | 41.5 | 3.9 | 8.0 | 1.7 | 2.5 | 4.7 |
| *Nunes et al. (2016)* | PA | * Includes SV in deficit and surplus calculation<br>* Includes Rural Settlements<br>* PRODES+ TerraCLASS data | Property (57, 890) CAR | 35.0 | 2.3 | 11.3 | 1.3 | 5.7 | 8.7 |
| *Brito (2017)* | PA | * Tests the impact of Land Tenure<br>* Includes SV in deficit and surplus calculation<br>* SV contribution explicit in results<br>* Includes Rural Settlements<br>* PRODES+ TerraCLASS data | Property (63,784) CAR, Incra Certified | 19.2 | 1.1*** | 4.7 | 0.7 | 4.9 | 6.7 |
| ***This study*** | **MT** | *** Includes SV in deficit and surplus calculation<br>* SV contribution explicit in results<br>* PRODES+ TerraCLASS data** | **Property (48,174) CAR** | **22.8** | **3.7****** | **1.4** | **0.8** | **0.6** | **1.8** |
| | **PA** | | **Property (96,141) CAR** | **23.9** | **1.9** | **5.4** | **0.9** | **3.3** | **6.0** |
| *Rajão e Soares-Filho (2014)* | BLA * | * Tests impact of land tenure and economic viability on surplus availability for CRA *(for comparison we selected numbers from relaxed scenario)*<br>* Does not include SV in surplus<br>* PRODES+ TerraCLASS data | Micro-watersheds | NA | 7.9 | 43.0 | 7.9 (exc. AM surplus) | 6.4 | 5.4 (2.33 inc. AM Surplus) |
| *Martini et al. (2015)* | BLA | * Includes SV in deficit and surplus calculation<br>* PRODES+ TerraCLASS data<br>* Includes Rural Settlements | Municipalities | NA | 14.1 | ? | ? | ? | ? |
| *Freitas et al. (2017)* | BLA * | * Tests regulative frameworks for an optimal conservation and social equity scenario<br>* Forest balance, calculating deficits and surpluses<br>* Does not include SV in deficit and surplus calculation | Property CAR (80%), Simulated (20%) | NA | 3.5 | 28.1 | 9.4 | 10.7 | 3.0 |
| ***This study*** | **BLA** | *** Includes SV in deficit and surplus calculation<br>* SV contribution explicit in results<br>* PRODES+ TerraCLASS data** | **Property (255,244) CAR** | **69.7** | **5.9<br>4.6******* | **12.0** | **3.6** | **2.6** | **3.3** |

# Chapter IV

# Enhancing synergies for effective large-scale forest restoration in Mato Grosso, Brazilian Amazon

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# Abstract

Widespread deforestation in Mato Grosso (MT), Brazil, has produced negative environmental outcomes of global reach. Now, MT has committed to partly offset this impact through the allocation of 2.9Mha of forest recovery statewide, motivated by the expectation of the Brazilian Forest Code (BFC) enforcement. Using an integrated approach, we mapped priority areas for allocating a fraction of MT recovery target to eligible public lands (1Mha) and private lands with forest deficits according to the BFC (1Mha). We estimated the costs and benefits of forest recovery for four scenarios, built varying the relative importance of four key criteria (habitat enhancement, carbon storage, opportunity costs of agriculture and likelihood of natural forest regeneration). Solely prioritizing habitat enhancement (scenario B100) improved the representation of species with highly disturbed habitats, while leveling the weights across criteria (scenario EW) yielded uniform enhancement across species. Conversely, EW costed 23% less than B100 (US$1.9 billion by 2030) and sequestered 27% more carbon (114.7 $TgCO_2e$ by 2030). In all scenarios, private properties were key to enhance intensively deforested habitats, while restoration in public lands was more effective in reducing costs and mitigating carbon. Our results stress that the BFC enforcement is crucial for increasing species representativity in highly deforested areas, but also revealed opportunities for cost-effective restoration across the state.

## 1. Introduction

Tropical forests biodiversity and carbon stocks are under immense pressure from rapidly expanding agricultural frontiers (Gibbs et al. 2010). In this regard, the Brazilian state of Mato Grosso (MT), located in the southeastern edge of the Brazilian Amazon has been on the spotlight for several decades. Occupation of MT was intensified in the 1970s, but the soybean boom in the mid-1990s made the state the largest exporter of agricultural commodities in Brazil (IBGE 2017; Nepstad et al. 2006). Such intense development came at great cost and, in only a few decades, the Amazonian share of MT had lost 16.9 Mha of old-growth (OG) forests, mostly to pasture and cropland expansion (INPE 2014c). A mix of policies aiming to stop deforestation followed the peak of forest loss in 2004 (Gibbs et al. 2015; MMA 2012; Nepstad et al. 2014), leading to a 91% reduction of clear-cut rates in MT by 2014, relative to 2004 (INPE 2014b). However, even with the recent progress, MT is still the second largest deforester in the Amazon (behind the state of Pará) as several active deforestation frontiers coexist with a modernized agricultural sector.

One of the major environmental governance challenges ahead of MT is the enforcement of the Brazilian Forest Code (BFC) (Brasil 2012). Enforcing the BFC has the potential to slow down deforestation and encourage restoration (Garrett et al. 2018). The BFC requires the conservation of up to 80% of forests in private properties of the Amazon biome as legal reserves (LRs) (Brasil 2012). Recent assessments estimated up to 8.6Mha of LR deficit in MT, the largest deficit amongst Brazilian states (Micol et al. 2013; Soares-Filho et al. 2014). To comply, where minimum criteria are not met, landholders may restore forests on-site or compensate LR deficits off-site. In this context, the state government committed to restore 2.9Mha of forests statewide (as a part of an agri-environmental development strategy called PCI – Produce, Conserve and Integrate), and 1.9 Mha of this commitment should take place via LR deficit recovery (PCI 2015). Such commitment represents 23% of the 12 Mha national restoration target (as established by the *Plano Nacional para Recuperação da Vegetação Nativa*, PLANAVEG, acronym in Portuguese) and 60.4% of the 4.8 Mha planned to be restored in the Amazon (Brasil 2017a).

The alignment between state, national and international (e.g., Aichi Targets, Bonn Challenge, and New York Declaration on Forests) restoration targets with the BFC enforcement could create a momentum for forest restoration, alleviating biodiversity loss and increasing carbon sequestration in MT (BonnChallenge 2017; Brasil 2015; Chazdon et al. 2017). However, the challenge of allocating nearly 3.0 Mha of forest recovery is not a simple one. Unlike punctual ecological restoration ventures, to be long lasting and effective, large-scale forest restoration requires a change in the rationale behind territorial planning, towards stakeholder's active engagement and the identification of opportunities for an intelligent allocation of resources (Holl 2017). Experts call for the need of a Forest Landscape Restoration (FLR) approach to identify areas of synergy between ecosystem services enhancement and livelihoods support. Such landscapes should be resilient enough to enable low cost restoration interventions and yet, land competition must be low, to avoid deforestation leakage and attract social actor's engagement (Chazdon and Guariguata 2016; Latawiec et al. 2015).

Like reserve selection frameworks (Schroter et al. 2014), FLR prioritization combines biophysical, socio-economic, and political criteria to identify prime areas that minimize trade-offs between ecosystem services and competing land uses across space, time and different scenarios (Gourevitch et al. 2016; Tobon et al. 2017), possibly accounting for climate change projections and uncertain land use trajectories (Zwiener et al. 2017). Carbon enhancement, erosion control, and biodiversity conservation are recurrent goals of forest restoration (Schulz and Schröder 2017). Biodiversity is often addressed by models that optimize connectivity, habitat quality and availability, using implicit (land use and cover maps) or explicit information on biological features distribution at many levels (umbrella species traits, species distribution models, species interactions, or genetic diversity information) (McRae et al. 2012; Molin et al. 2018; Proft et al. 2018; Thomson et al. 2009). Focusing on Brazil, recent studies mapped cost-effective areas for forest restoration allocation accounting for the spatial distribution of LR deficits (Molin et al. 2018; Nunes et al. 2017; Oakleaf et al. 2017) and considering different BFC implementation strategies (Kennedy et al. 2016b; Stefanes et al. 2016; Strassburg et al. 2019). However, no study has simulated FLR outcomes across large scales using actual property boundaries as allocation units, an exercise that would bring much needed realism to inform restoration policies and planning.

In this paper, we focused on the Amazonian share of MT to address the FLR challenge of distributing a fraction (1.0 Mha) of the state's restoration target in private lands (intra-property allocation) with legal reserve deficits. We chose to only allocate a fraction of MT's restoration target destined to legal reserves because of the state's tripartition in three biomes (i.e., Amazon, Cerrado and Pantanal). Our objective was to assist FLR planning across scales: from the identification of regional hotspots to the allocation of restoration in rural private properties with LR deficit. Even though MT's restoration strategy focuses on private lands, to compare the costs and benefits of restoration in lands under different jurisdictions we also allocated the same amount of restoration (1.0Mha) to eligible public lands. We considered four criteria, known to be relevant for large-scale forest restoration and which can be divided in two groups: (i) restoration feasibility (i.e., likelihood of natural forest regeneration and land opportunity costs) and (ii) forest functions enhancement (i.e., carbon storage and habitat suitability). We explored four scenarios, produced varying the relative importance of criteria in our framework with the objective to anticipate the variation in costs and benefits outcomes of an imbalance in setting priorities for FLR. Specifically, we addressed the following questions:

(1) How do costs and benefits of allocating forest restoration to priority areas vary across space and under different scenarios?

(2) How do costs and benefits of forest restoration in private and public lands compare?

(3) Which areas show higher overlaps and conflicts between costs and benefits for different scenarios?

# 2. Material and methods

## 2.1. Study area

MT is the southernmost state of the Brazilian Legal Amazon, covering 90.6Mha, of which 48.0Mha are within the Amazon Biome (MMA 2017a) (Supplementary Figure IV-1a). Vegetation ranges from semi-deciduous to evergreen rainforests, typical of the transition from savanna ecoregion (i.e., Cerrado, in the south) to the tropical rainforest (north) (IBGE 2004). By 2014, 31.1 Mha (66.5%) of OG forests were standing and 3.8 Mha were under different stages of forest recovery (INPE 2014b; INPE 2014c). Protected or indigenous lands contained 33.8% of OG forests (INPE 2014b). Remaining forests stocks were either located in undesignated public lands or unknown tenure (21.9%), settlement projects (2.0%) or private properties (42.1%) (own calculations based on data compiled for this study). Among private properties, large-scale farms (>1,000 hectares properties) concentrated 80% of forests (Richards and VanWey 2016).

OG forest remnants are heterogeneously distributed across MT (Supplementary Figure IV-1a). At the south-central portion of the state lie the older and mostly inactive deforestation frontiers, on plateaus dominated by capital-intensive mechanized agriculture developed by large farms; landscapes are highly fragmented. In eastern MT, large swaths of well-preserved forests are encompassed by the Indigenous lands *Parque do Xingu* and *Jarina-*

*Capoto,* with borders surrounded by private lands; these fragile areas concentrate most headwaters of the Xingu River and have been strongly impacted by deforestation. At the north, widespread deforestation due to cattle ranching and mining takes place in lands mostly unsuitable for mechanized agriculture. Finally, in northwestern MT lies the most recent and active deforestation frontier in MT, characterized by small clearings open in fishbone patterns (Davenport et al. 2017)

## 2.2. General approach

Our approach to allocate FLR to priority areas in MT (Figure IV-1) can be described by six steps. *First*, we defined four criteria, representing two overarching goals of forest restoration: (a) restoration feasibility, guided by the (i) likelihood of natural forest regeneration and (ii) opportunity costs of forest restoration; and (b) forest functions enhancement, guided by the potential for (iii) habitat enhancement for multiple species and (iv) carbon storage. *Second*, the four criteria were combined using a prioritization algorithm, which produced a hierarchical ranking of restoration value over the entire landscape. *Third*, we performed a sensitivity analysis, systematically reducing the relative importance of habitat enhancement relative to the other three criteria which resulted in four prioritization scenarios. *Fourth*, we identified the opportunities for forest restoration allocation in private lands (i.e., identification of private rural properties with a LR shortage) and public lands. *Fifth*, we proceeded to allocate the restoration target to private and public lands; allocation to private properties involved sorting and selecting private properties with LR deficit (i.e., opportunities for FLR) according to the restoration rankings produced by each scenario. *Sixth*, we estimated and compared the financial costs of restoration with the benefits for forest function enhancement under each prioritization scenario.

All data were processed and analyzed using the South American Albers Equal Area Conic projection and regular grids with 1 ha (100x100 meters) cell size. Supplementary Table IV-1 presents the variables used in our model.

## 2.3. Forest restoration criteria

### *2.3.1. Natural forest regeneration likelihood*

FLR has greater chances of success at lower implementation costs if allocated to lands where regeneration would occur naturally, due to a combination of socio-economic and environmental factors that makes these lands marginal for maintaining agriculture and yet appropriate for forest recovery (Holl 2017; Lamb et al. 2005; Molin et al. 2018). Following this assumption, we used a spatially explicit land change model implemented with Dinamica Ego v.4, (Soares-Filho et al. 2009), to estimate the probability of natural forest regeneration, here represented by the land cover transition from agriculture (i.e., pasturelands or croplands) to secondary vegetation. Dinamica Ego uses Weights of Evidence (WoE), a Bayesian conditional probability method, to calculate the spatially distributed likelihood of a given land use transition (Soares-Filho et al. 2010). The transitions from agriculture to secondary vegetation were identified by crossing land use maps for the years 2012 and 2014, corresponding to initial and final time-steps, respectively (INPE 2014b; INPE 2014c).

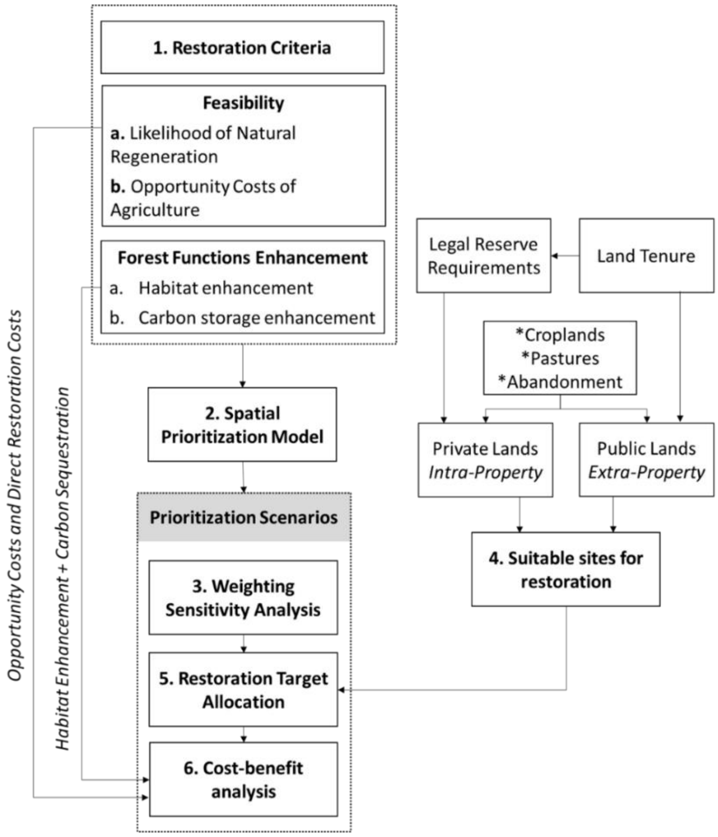



**Figure IV-1. Methodological workflow.** Conceptual framework describing the main methodological steps taken to identify priority areas for forest restoration and allocate the state-level restoration target in Mato Grosso (MT) under different scenarios.

Probabilities were calculated based on the influence (weights) of a set of independent biophysical (i.e., soils, slope or distance to rivers) and socio-economic (i.e., distance to forest edges, settlements and infrastructure and land use restrictiveness) variables on forest regeneration and agricultural land abandonment in the tropics (Molin et al. 2017; Monteiro et al. 2018; Teixeira et al. 2009). See Supplementary Text IV-1 for a description of model calibration, Supplementary Figure IV-2 to 7 depict weights for individual variables included in the model, and Supplementary Figure IV-8 for the resulting forest regeneration likelihood map.

### *2.3.2. Opportunity costs*

Forest restoration is perceived as a cost by landholders, proportional to the forgone profits (i.e., opportunity costs) of an alternative land use, such as annual agriculture or cattle ranching (Stefanes et al. 2016). Therefore, high opportunity costs are expected to incur in less engagement in forest restoration by landholders. We used prices of land under pasture and annual agriculture use as an indicator of the opportunity costs from setting land aside for restoration (FNP 2012; Rajão and Soares-Filho 2015). Values were obtained from several localities and are representative of microregions (i.e., planning units representing an assemblage of municipalities with similar economic productive sectors) intersecting MT in the Amazon Biome (IBGE 2014). To make land price estimates continuous in space, we took the average of land prices within each microregion weighted by the area used as pasture and agriculture in the respective microregion (Supplementary Figure IV-9) (Cai et al. 2016; Strassburg et al. 2019); for calculating the weighted average we used TerraClass data from the year 2014 to assess land use area (agriculture and pasture) (INPE 2014c). From this operation we obtained a surface of average land price values per microregion (Supplementary Figure IV-9). All values were corrected to 2018 for inflation and converted to US currency using an exchange rate of 2018 of 1 US$ = R$ 3.83.

### *2.3.3. Habitat enhancement for multiple species*

We relied on a functional landscape connectivity approach to identify the areas where forest restoration allocation would provide most habitat improvement for multiple species in our study area. Using terrestrial mammals geographical range information, obtained from the IUCN Red List of Threatened Species (IUCN 2018) we identified 206 terrestrial mammal species present within Amazonian MT; from this group we selected 62 species of non-flying mammals with available information about their specific home range, using several datasets and references as sources, among other references (Supplementary Table IV-2). The home range is a species trait that indicates the area in which an animal lives and moves periodically, frequently used to support conservation and restoration of habitat connectivity planning (Bingham and Noon 1997; Fauvelle et al. 2017; Piquer-Rodríguez et al. 2015). Previous studies have used habitat requirements from medium to large sized non-flying mammals as ecological indicators, due to their strong response to habitat patch size and isolation, and to forest expansion (Arevalo-Sandi et al. 2018; Soares-Filho et al. 2006).

Next, we created habitat degradation level maps for each species by crossing the geographical range layers with a reclassified land cover map (TerraClass) (INPE 2014c) and assigned scores indicating habitat quality based on the level of degradation associated with each land cover (Supplementary Table IV-1). Areas undergoing forest regeneration were given higher scores than areas under active agricultural use, depending on the duration of the abandonment period. Previous literature positively correlates the length of the abandonment period with the suitability for natural restoration and support of biodiversity (Arevalo-Sandi et al. 2018; Chazdon and Guariguata 2018). After ~5 years of abandonment, chances of re-clearance decrease, which makes it a good indicator for long term restoration success (Müller et al. 2016b; Schwartz et al. 2017). Areas occupied by annual agriculture were given low scores to offset prioritization towards areas with less intensive land use history. This is because highly mechanized annual agriculture subjects land to procedures such as plowing, tillage and heavy use of chemical defensives, making natural revegetation unlikely due to both high opportunity costs of agriculture and direct costs of input-demanding active restoration projects (Arroyo-Rodriguez et al. 2017). Using a regular grid (25 km2 cell size) we stacked the geographical range layers of our species of choice and found that the number of species was well distributed in MT, varying from 34-46 species out of maximum of 62 possible. However, species were not equally represented across the study area, as well as not equally affected by deforestation (Supplementary Table IV-3).

### *2.3.4. Potential carbon storage*

We used the forest live biomass map produced by the Brazilian Third Emission Inventory (BEI) (Brasil 2016b) as an indicator of the spatial heterogeneity of the carbon storage potentially offered by forest restoration. Aboveground biomass (AGB) was estimated for the BEI using diameter at breast height information from pristine vegetation areas inventoried by the RadamBrasil project, which were applied to allometric equations calibrated for the Amazon vegetation types and spatialized using interpolation (Brasil 2016b) (Supplementary Figure IV-10a). Belowground biomass (BGB) was obtained by BEI from the scientific literature, data collected from the RadamBrasil project or using AGB expansion factors and spatialized using interpolation (Supplementary Figure IV-10b).

## 2.4. Forest restoration prioritization

We used Zonation v. 4.0 (Moilanen et al. 2014) to prioritize areas for restoration in MT. Zonation assumes integral landscape conservation as the optimal solution and iteratively removes grid cells while minimizing the global loss of biological value, producing a nested rank (0-1) of the landscape (Moilanen et al. 2005). Hence, cells removed first have lower biological value than those removed last, and such rank can be used to strategically allocate conservation actions.

Zonation was originally conceived to prioritize conservation of current habitats but has been increasingly used to address restoration problems (Budiharta et al. 2016; Thomson et al. 2009; Zwiener et al. 2017). Like Thomson et al. (2009), we worked around this limitation assuming that all suitable areas had been restored and proceeded with the iterative cell removal until only the current forest configuration was remaining. We used a hierarchical removal mask of extant forest to ensure that OG forests would be removed last and would not interfere with the ranking of candidate areas for restoration (Thomson et al. 2009). This method also privileged the prioritization of pixels close to OG forests edges, naturally more prone to natural regeneration. We "masked out" areas that were not viable for restoration (urban areas, water bodies or non-forest ecosystems) using the TerraClass land cover product (INPE 2014c).

In Zonation, rank values are assigned according to the input features and the chosen cell removal rule (Monteiro et al. 2018). Features to be enhanced (i.e., multi-species habitat, carbon storage and likelihood of forest regeneration) were given positive weights and restoration constraints (opportunity costs) were given negative weights (Di Minin et al. 2017; Moilanen et al. 2011). Each of the four criteria was entered as a group and given a weight in Zonation, so that their absolute values summed to 100. In the case of the multi-species habitat suitability criterion each one of the 62 species accounts for the 0.01613 fraction of the group's weight. Prior to the ranking, each species' habitat quality map and home range (Supplementary Table IV-1 and Supplementary Table IV-2) information were pre-processed by Zonation 4.0, using the Distribution Smoothing function (Moilanen and Wintle 2006) to enhance connectivity between high-quality habitat patches and identify species specific suitable habitats given the respective species home range requirement. Cells were removed using the Additive Benefit Function, in which the value of a grid cell is given by the "sum over species-specific values, and species-specific value is an increasing function of representation" (Moilanen 2007, p. 571). Therefore, cells are removed in an order that minimizes global loss of features value at each iteration.

## 2.5. Forest restoration prioritization scenarios and sensitivity analysis

Ideally, sites showing high potential for ecological enhancement should be assigned full priority for restoration, however, they are not always feasible candidate areas from the socio-economic standpoint (Arponen et al. 2010). Therefore, we conducted a sensitivity analysis and shifted the dominance of the habitat variable (Figure IV-2) in our spatial prioritization framework. Our aim was to understand how varying the underlying objectives of forest restoration offset the ranking and impacted expected costs and benefits.

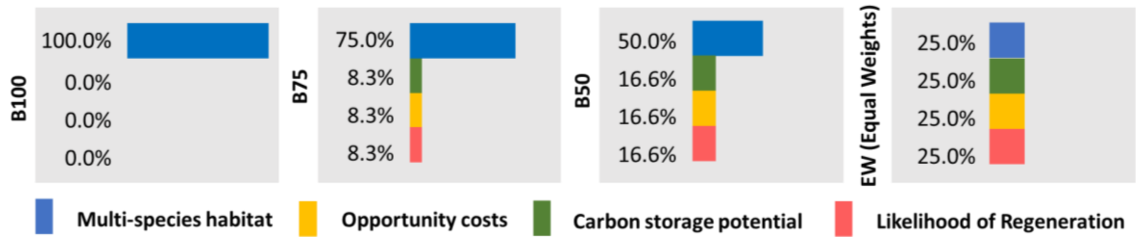



**Figure IV-2. Criteria and combination of weights.** Scenario B100 fully prioritizes (100%) habitat enhancement for multiple species, subsequent criteria progressively level weights among criteria.

We incrementally altered the weights attributed to habitat suitability from 100 (full habitat suitability prioritization) to 25 (equal weights) in the modeling framework, while increasing the total weight of the three other criterion from 0 to 75, resulting in four prioritization scenarios (Figure IV-2). We named these scenarios as B100, B75, B50 and Equal Weights (EW).

## 2.6. Forest restoration target allocation

In this study, suitable sites for FLR are any areas equal or larger than 1.0 ha which underwent forest clear cut (INPE 2014b) and were covered by either pasturelands, croplands or forest regeneration by 2014 (Figure IV-1, Supplementary Figure IV-1, Supplementary Table IV-1). We allocated forest recovery outside (1.0Mha) and inside (1.0Mha) private properties in Amazonian MT based on each prioritization scenario. This extent (1.0Mha) consists of 52.6% of the state level restoration target for LRs (1.9Mha) and 68.9% of the total state restoration target (2.9Mha). Allocating an equal amount of restoration to private and public lands allowed us to compare the costs and benefits of prioritizing extra or intra property allocation.

Over 4.4 Mha of suitable candidates were identified for extra-property target allocation spanning over previously clear-cut areas located in public, indigenous or lands with unknown tenure (deemed as public lands for the purpose of our study).

For the intra-property FLR allocation, we restricted eligibility to private properties showing LR deficit, i.e., a legal demand for restoration according to the BFC, as calculated by Hissa et al. (2019) (Supplementary Text IV-2, Supplementary Figure IV-11, Supplementary Table IV-1). The authors (Hissa et al. 2019) identified 3.4 Mha in LR deficits, here computed as suitable candidates for intra-property allocation of forest restoration. We individually overlaid the rank maps on a layer of properties with deficits and selected the top fraction of cells from the rank needed to offset deficits from each non-compliant landholding. Second, we inputted all simulated restored LR areas - with unique property identifiers - back into Zonation, together with the respective ranking maps and ran prioritization analyses using the *Planning Units* function (Di Minin et al. 2017; Fagundes et al. 2018). Using this function, each unit (i.e., restored LR) was removed entirely at each iteration, in an order that minimized global loss of value in the rank map. This process yielded a ranking of target properties for a state-wide restoration program for each scenario, based on the BFC demand for LR deficit offsetting. Properties were then selected until the 1.0 Mha target was fulfilled to create a mask of restored LR used to compare costs and benefits across model setups (see section 2.7).

## 2.7. Calculating the costs and benefits of forest restoration

If driven by the PCI strategy, forest restoration will not take place immediately, but rather in incremental steps until 2030, the anticipated year for the program closure. Therefore, to compare restoration financial costs with benefits for forest functions we considered that forest expansion would be allocated at a linear rate across 11 years, between 2019 and 2030. To do so, we segmented the allocated restoration target into 11 equally sized sub-regions of $\approx$ 0.09 Mha in private properties and 11 equally sized sub-regions of $\approx$ 0.09 Mha in public lands, with each sub-region representing an annual increment in restored forest area. The restoration target segmentation followed a minimum to maximum monetary costs approach; areas showing lower monetary costs were set aside for restoration at the beginning of the period and areas showing higher monetary costs were restored towards the end of the period, 2030.

Here, monetary restoration costs equal to the sum between (i) opportunity costs of agriculture and (ii) direct costs of restoration. First, to calculate the opportunity costs, we used the land prices included in the prioritization analysis to derive an estimate of annual profits from agriculture foregone for forest restoration. We followed the approach by Cai et al. (2016), who divided land prices (*see section 2.3.2. Opportunity costs*) by ten to approximate profits obtained from annual land leases. We included foregone profits in the total monetary costs of restoration for private properties, but also for undesignated public lands or with unknown tenure which are often under agricultural use. This is because implementing restoration activities in these candidate areas will either involve compensating rightful landholders or increasing law enforcement to end illegal land use, which also incurs in financial expenses. We did not add opportunity costs of restoration taking place in protected or indigenous or lands.

Direct restoration costs vary widely with the required restoration strategy, which in turn depends on the level of degradation and ecological resilience of the ecosystem (Bullock et al. 2011). Like recent studies, we interpreted the likelihood of forest regeneration as an indicator of the ecosystem capability to naturally recover from previous disturbances in marginal lands (Brasil 2017b; Molin et al. 2018; Nunes et al. 2017). Therefore, areas showing low probability of regeneration would likely need high-cost inputs for revegetation (e.g., use of fertilizers, active seedlings planting) whereas high probability areas would demand less input (e.g., passive restoration, with costs restricted to fencing, monitoring and minor interventions) (de Groot et al. 2013). Therefore, similar to Molin et al. (2018) and Nunes et al. (2017) we sliced the likelihood map into four categories using increments of 25%, representing a gradient in the demand for input in native restoration projects, and assigned costs relative to each restoration method as reported by Timotheo et al. (2016) (Supplementary Table IV-4). We did not include costs which may also be involved in implementing large-scale forest restoration, such as planning, creation of infrastructure or law enforcement. We applied a 10% annual discount rate to obtain present values for 2019 for both opportunity costs and direct forest recovery costs over the 11 years' timeframe (2019-2030) (Molin et al. 2018; Rajão and Soares-Filho 2015).

To estimate restoration benefits, we compared high-quality habitat gains between scenarios for each species inside and outside properties. We examined the percent increase in species distributions proportions after allocating 1 Mha of restoration to private lands and 1Mha to public lands according to each prioritization solution. To do so, we ran the B100 prioritization scenario in Zonation replacing candidate cells targeted by each scenario with high-quality habitat (i.e., score=1). We then compared the relative increase between the original distribution proportion and the new simulated distributions for our scenarios. We worked under the assumption that habitat quality would be fully recovered on restored cells (i.e., ecological functions are immediately recovered).

Secondly, we evaluated the cost-effectiveness of carbon enhancement (i.e., monetary costs per unit of carbon dioxide – USD/Ton.CO2) across scenarios. Here, carbon enhancement is understood as the sum between (i) the amount of carbon sequestered by forest regrowth from 2019 onwards with (ii) carbon stocks previously stored in regrowing forest in the case they are selected as priority. Carbon sequestration was estimated annually using the 11 steps increments from 2019 to 2030 as time lag indicators. We multiplied the forest biomass (above and belowground) values spatially distributed across restoration increments by an annual biomass accumulation rate of 1.2%, as estimated by Lennox et al. (2018). To calculate the amount of carbon previously stored in secondary vegetation areas selected as priority in each scenario in carbon cost-effectiveness calculations. The land cover map allowed us to track the secondary vegetation age between 2004 and 2014, to which we added four years of carbon accumulation to add up to 2018; we arbitrarily assumed no forest re-clearance occurred after 2014. Restoration costs were then divided by the amount of carbon stored in the restored landscapes by 2030 to estimate the most cost-effective scenario for carbon sequestration. A carbon fraction of 0.5 was applied to convert biomass values to carbon density.

# 3. Results

## 3.1. Priority areas and allocation of forest restoration

Leveling the weights assigned to criteria caused important shifts in high and low priority areas across scenarios (Figure IV-3). In the B100 scenario high priority values were strongly concentrated in southwestern MT (Figure IV-3a, Supplementary Figure IV-12c) but also distributed near OG forest borders, whereas in the EW scenario high values shifted towards a more contiguous distribution across north and northwestern MT (Figure IV-3b). Local differences in restoration allocation were also observed across scenarios (Supplementary Figure IV-12a-c). The allocation of restoration to the B100 priority map yielded patches of restored forest with rounder borders in comparison to the allocations to priority maps from other scenarios, which formed lengthier patches with a strong influence of riparian margins due to the inclusion of the forest regeneration likelihood criterion (Supplementary Figure IV-12 ).

Shifts in restoration priorities between scenarios influenced the selection of private properties for LR deficit offsetting across MT (Supplementary Figure IV-12a-c). This produced more dissimilar patterns of simulated restoration among scenarios for the intra-property allocation in comparison to the extra-property allocation, which showed smoother shifts in restoration distribution between the four scenarios (Supplementary Figure IV-12a-c). This is because the planning units' function (see section 2.6) used to allocate the private property restoration target, iteratively removes candidate properties with simulated LRs aiming for a minimum aggregate loss in global restoration priority value across the landscape. Hence, if one or more pixels are ranked with high priority, but are located within the boundaries of a property with a low overall restoration value, such pixels will not be allocated for restoration, since the property will likely be excluded in detriment of other properties with higher total restoration value. Conversely, the allocation of restoration in public lands was not subjected to this constraint.

Target allocation overlap was strongly concentrated in northwest MT (Supplementary Figure IV-13). One million hectares were mapped as priority by all scenarios, while 0.8Mha where selected by only one. The four allocation scenarios showed an average overlap of 69.4% (16.8% SD) for private properties and 75.3% (12.0% SD) for restoration allocation to public lands. Overlap increased as the weights assigned to criteria were evened, surpassing 90% of agreement between the B50 and EW scenarios for both public and private lands allocation (Supplementary Table IV-5). The B100 scenario showed the least spatial agreement with other allocated targets.

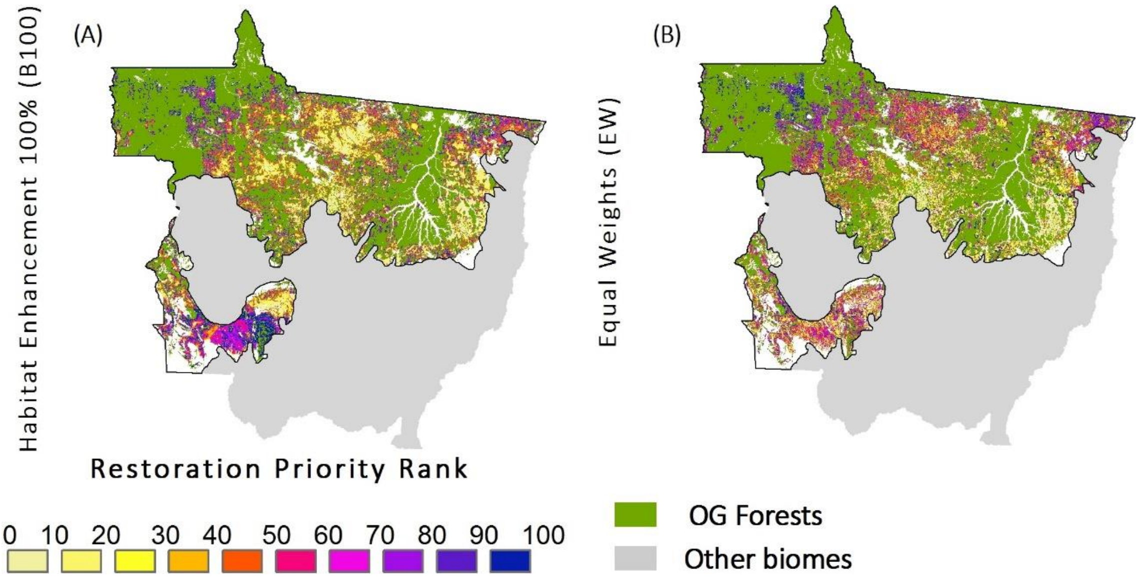



**Figure IV-3. Forest restoration priority rank maps for two selected scenarios.** In "B100" (A) only habitat enhancement (for the 62 species analyzed by this study) guided prioritization while in "EW" (B) the four criteria (i.e., habitat enhancement, carbon storage potential, likelihood of forest regeneration and opportunity costs of agriculture) were considered and received equal weights. Rank values for restoration vary between 0 (no value) and 100 (maximum value) for each scenario. Non-ranked cells within the Amazon biome are either represented in white (non-forest land cover) or in green color (old growth "OG" forests).

## 3.2. Benefits of forest restoration for habitats enhancement and carbon storage

Differences in the average increase in species representation was almost nil between the scenarios (6.6%) but showed a decreasing variance from B100 to EW (Table IV-1).

Conversely, habitat improvement per species was not uniform across scenarios (Supplementary Table IV-6 to 9); for example, distribution increase reached 29.0% and 12% for the species *Mazama gouazoubira* in scenarios B100 and EW, respectively (Figure IV-4a-b). Scenario B100 prioritized underrepresented species, more affected by deforestation (Figure IV-4a) occurring at the southern border of the Amazon biome. The EW scenario performed more evenly across species, and better than B100 at increasing the representation of species occurring in northwestern and central MT, not necessarily those more threatened (Figure IV-4b, Supplementary Table IV-6 to 9). Species with large home range benefited less from restoration (Figure IV-4a-b), with a slightly increasing performance from B100 towards the EW scenario. Intra-property restoration was responsible for most increase in the distributions of poorly represented species, while well represented species were favored by extra-property forest recovery (Supplementary Figure IV-14; Supplementary Table IV-6 to 9). The same pattern was present across scenarios, but the difference between intra and extra-property increments for species distributions decreases from B100 to towards the EW scenario (Supplementary Figure IV-14; Supplementary Table IV-6 to 9). This shows that the choice of different properties for LR deficit compensation by each scenario had the largest impact on the decrease in performance of the habitat enhancement for poorly represented species, as opposed to the extra-property restoration allocation.

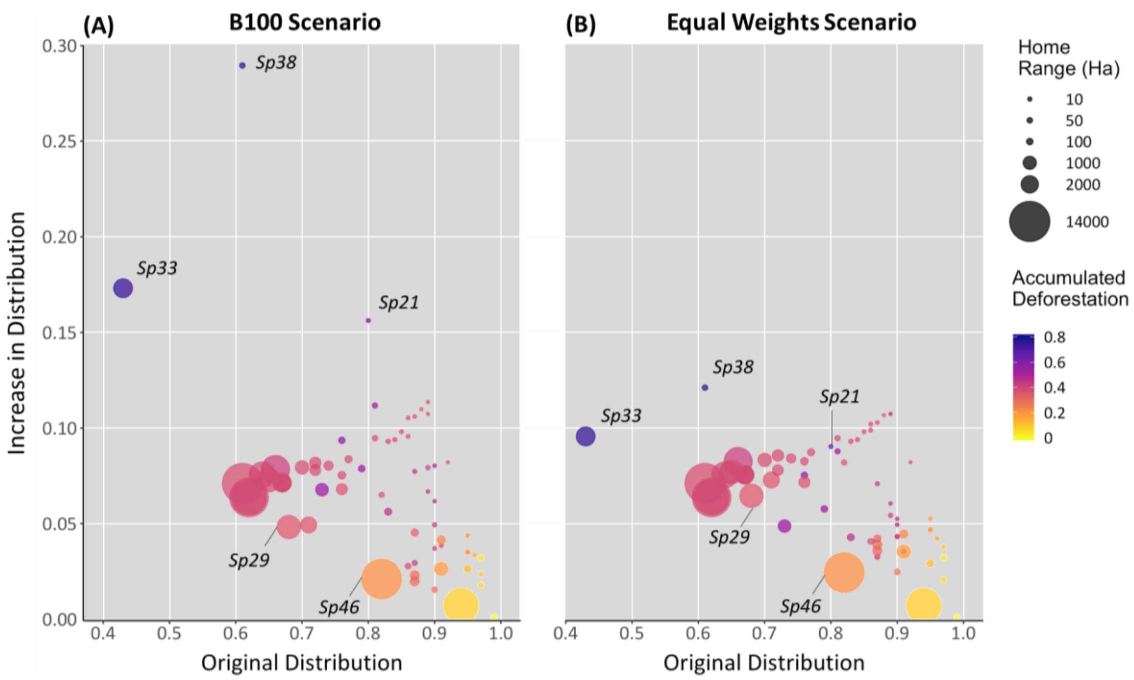



**Figure IV-4. Percent increase in original species distributions.** in high quality habitat after allocating 2 Mha of restoration following (A) B100 and (B) Equal Weights scenarios. Sphere sizes refer to the home-range of the species measured in hectares. Color gradient indicates the proportion of the original forest cover in the species geographic range lost to deforestation.

**Table IV-1. Restoration benefits for habitat enhancement and carbon mitigation.** Total for private properties (intra-property/IP), public lands (extra-property/EP) and total. EW = Equal Weights.

| | **Average Habitat Gain (%)** | | | **Carbon Mitigation ($Tg.CO_2e$)** | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | | | Carbon Uptake (2019-30) | | | Secondary Vegetation protection | | |
| | IP | EP | Total | IP | EP | Total | IP | EP | Total |
| **B100** | 3.9 (3.1) | 3.2 (2.0) | 6.9 (4.3) | 31.2 | 33.8 | 65.0 | 21.4 | 28.4 | 49.8 |
| **B75** | 3.6 (2.5) | 2.8 (1.4) | 6.6 (3.5) | 32.9 | 34.9 | 67.8 | 30.6 | 39.0 | 69.6 |
| **B50** | 3.5 (2.2) | 2.8 (1.2) | 6.5 (3.0) | 33.0 | 35.0 | 68.0 | 31.9 | 44.4 | 76.3 |
| **EW** | 3.4 (1.8) | 2.7 (1.1) | 6.4 (2.7) | 33.2 | 34.9 | 68.0 | 31.9 | 46.1 | 78.1 |

The restoration of 2.0 Mha of forests in MT based on scenario B100 would mitigate 114.7 $TgCO_2e$ by 2030 (Figure IV-5), while prioritizing the four criteria equally (EW) could rise the mitigation potential to a maximum of 146.1 $TgCO_2e$, a 27.3% increase. Most increase would already be achieved by the intermediate scenarios B75 (19.9%) and B50 (25.7%) and most of the variation is associated with carbon previously stored in secondary vegetation areas selected as priority (Figure IV-5, Table IV-1). Restoration allocations guided by scenarios showing higher overlap with areas covered by secondary vegetation by 2014 (Supplementary Figure IV-15) presented higher carbon mitigation potential. Forest restoration in public lands would mitigate 20.4% (3.1% SD) more carbon than LR restoration (intra-property) (Figure IV-5). This difference is less prominent in B100 compared to the Equal Weights scenario, as the share of secondary vegetation areas selected as priority increases more in public lands than inside private properties across scenarios (Supplementary Figure IV-15).

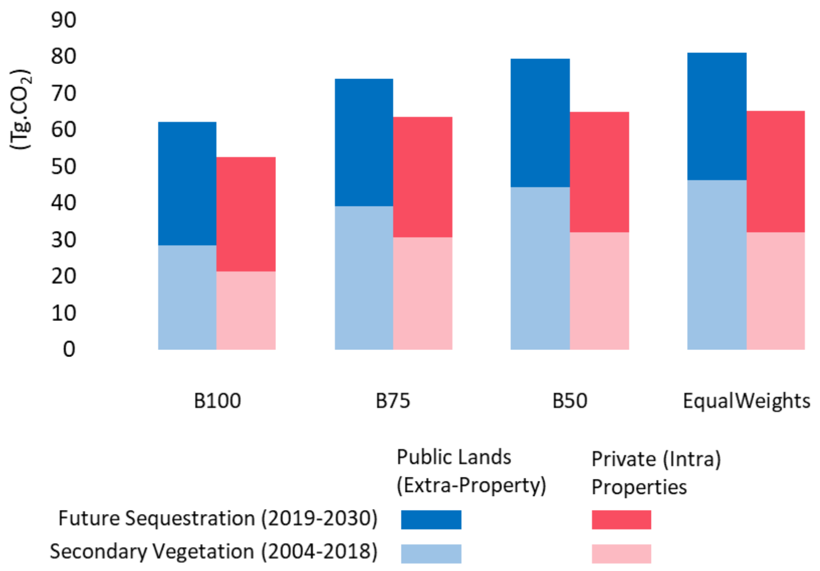



**Figure IV-5. Carbon (CO2) mitigation by selected areas for regeneration by 2030** in public (blue bars) and private lands (red bars). Carbon offsets are divided between carbon accumulation in secondary vegetation (dark toned bars) and carbon sequestered between 2019-2030 in lands selected as priority for restoration (light toned bars).

## 3.3. Costs of forest restoration and cost-effectiveness of carbon mitigation

Scenario B100 incurred the highest direct and opportunity costs, totaling 2.6 billion USD. Scenario B75 would incur in 23.6% less costs, 4% more than the economy of 27.2% achieved by the EW scenario, which totaled 1.9 billion USD (Figure IV-6, Table IV-2, Supplementary Figure IV-16). In scenario B100, forest restoration in public lands was 29.8% less costly than promoting LR forest recovery, the largest difference between private and public land restoration costs among scenarios. The difference drops to 13.3% for B75 and remains around this level in the other scenarios, mostly due to a sharp reduction in direct restoration costs in B100 in the order of 0.4 billion USD compared to the EW scenario (Figure IV-5, Table IV-2).

**Table IV-2. Restoration costs and cost-effectiveness of the carbon mitigation potential.** Total values are presented for each scenario, as well as a distinction between private properties (intra-property/IP), public lands (extra-property/EP). Estimates are relative to 2019-2030. EW = Equal Weights.

| | Restoration Costs (USD Billion) | | | | | | Cost-effectiveness USD/Ton.$CO_2$e | | |
|---|---|---|---|---|---|---|---|---|---|
| | Direct Costs | | | Opportunity Costs | | | | | |
| | IP | EP | Total | IP | EP | Total | IP | EP | Total |
| **B100** | 1.2 | 0.8 | 2.0 | 0.3 | 0.3 | 0.6 | 29.0 | 17.2 | 22.6 |
| **B75** | 0.8 | 0.6 | 1.4 | 0.3 | 0.3 | 0.6 | 17.4 | 11.8 | 14.4 |
| **B50** | 0.8 | 0.6 | 1.3 | 0.3 | 0.3 | 0.6 | 16.5 | 10.5 | 13.2 |
| **EW** | 0.8 | 0.6 | 1.3 | 0.3 | 0.3 | 0.6 | 16.3 | 10.3 | 12.9 |

The EW scenario offered the most cost-effective carbon mitigation (Figure IV-7, Table IV-2). Each ton of $CO_2$ could be offset at the cost of 12.9 USD, a reduction of 42.8% in comparison with B100 (22.6 USD/ton/$CO_2$), the least feasible from the economic stance. Cost-effectiveness was also different in public and private lands. In B100, target restoration would cost 17.2 USD/ton/$CO_2$ in public lands while the same carbon offset costed 29.0 USD/ton/$CO_2$ when allocated to private properties with deficit. The difference in cost-effectiveness is reduced as the carbon and feasibility criteria gained importance in prioritization (Table IV-2).

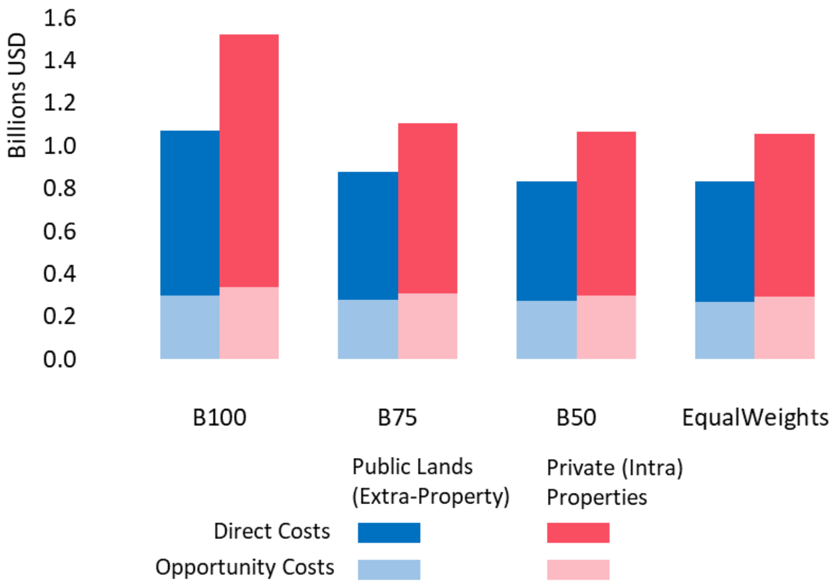



**Figure IV-6. Total restoration costs (USD billions) by 2030.** For each scenario, left-sided bars show costs of restoring of (1 Mha) forests in public lands (extra-property) and right-sided bars show intra-property restoration costs. Costs are divided between direct (dark toned color) and opportunity (light toned color) costs.

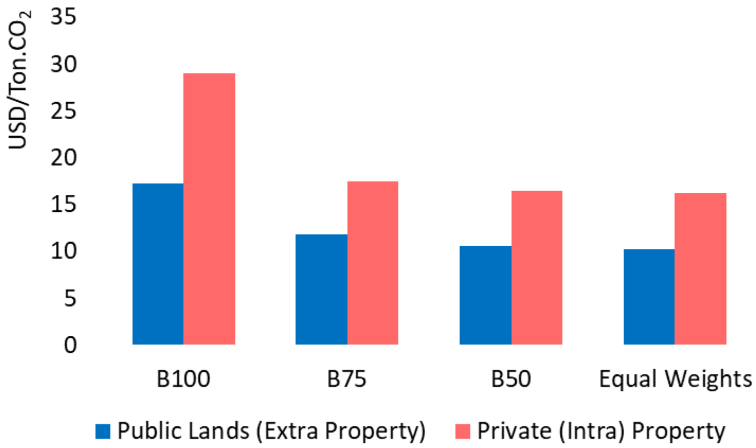



**Figure IV-7. Carbon mitigation cost-effectiveness for each scenario** (USD/ton/$CO_2$). Left-sided bars (blue shade) indicate cost-effectiveness of restoration allocation to public lands (1Mha) and right-sided bars (red shade) to private lands (1Mha). Values represent total cost-effectiveness by 2030.

## 4. Discussion

Few studies have applied prioritization methods to allocate restoration demands created by specific land-use legislation using actual property boundaries information (but see Kennedy et al. 2016). In this study, we addressed this gap, and identified prime areas for forest recovery in MT considering the restoration demand created by public policies (i.e., PCI strategy) under a BFC implementation scenario. Our approach integrates socio-economic and biophysical criteria and revealed trade-offs between feasibility and forest functions enhancement. On the one hand, when exclusively targeting biodiversity (B100) the habitats of poorly represented species (i.e., highly degraded habitats) could be substantially expanded. Habitat gains would be gradually abated at more cost-effective scenarios. On the other hand, most of the financial savings and carbon mitigation promoted by the increasing importance of feasibility criteria would already be achieved by intermediate scenarios, such as B75 and B50. This underscores the importance of spatial planning tools to the prioritization of restoration in areas where high trade-offs are expected.

By 2030, most of the economic advantages of the scenarios B75, B50 and EW were due to reduced direct restoration costs assigned to areas with a high likelihood of forest regeneration (Figure IV-6). Particularly, secondary vegetation areas were assigned the lowest direct restoration costs and were increasingly selected as cost-effective locations for forest restoration, especially in public lands (Supplementary Figure IV-15). Besides reduced direct costs, accounting for protection of the carbon previously stored by secondary vegetation as additional mitigation offered by restoration programs could increase restoration cost-effectiveness, partly financing FLR through payments for carbon sequestration (Olschewski and Benítez 2005). Moreover, the extent of secondary vegetation selected as priority by all scenarios and its potential for increasing FLR feasibility stresses the need for effective laws and policies for the protection of second-growth forests (Vieira et al. 2014). To support this discussion, further studies confirming in loco the functional viability of secondary vegetations and research on forest regrowth underlying processes are highly needed (Arroyo-Rodriguez et al. 2017; Carvalho et al. 2019a; Lennox et al. 2018).

The average increase in species representation was steady across scenarios, but we observed important differences in habitat enhancement between species. The concave form of the benefit function used in Zonation incurs in higher value losses when cells covering poorly represented species are removed. However, because it is an additive function, prioritization privileges a higher global average value for a solution, allowing for trade-offs between features (Arponen et al. 2005) in cases of conflicts with other criteria. This means, it may prioritize cells with high representation values for most features in detriment of cells containing high values for few, threatened, rare or endemic, features. Therefore, as we increased the weights of other restoration criteria, the average global increase in species distributions remained the same, but poorly represented species, located in unfavorable cells for carbon, opportunity costs and likelihood of regeneration, performed worse. In the future, if restoration focuses on one or more specific species, one solution to assure habitat enhancement for those could be to assign higher weights for each targeted species or condition allocation to a desired representation level.

The geographic range of species with very disturbed habitats overlaps areas with high density of private properties and fewer candidate areas for restoration in public lands. We demonstrated that, for this reason, recovering LR of private properties with deficits offers the main opportunity to significantly increase the habitats for underrepresented species in MT, with scenario B100 presenting the best performance (Supplementary Figure IV-14). A previous study by Kennedy et al. (2016) found that unconstrained allocation of restoration could bring higher benefits for connectivity than forest recovery restricted to private properties with LR deficits. Although we do not question the effectivity of spatially unconstrained restoration in reducing fragmentation, we did not test this approach because – at the early stages of a statewide restoration program – the high costs of forest restoration make voluntary FLR in compliant landholdings unlikely. Instead, we show that a careful selection of priority private properties for the compensation of LR deficits via restoration could be crucial to avoid redundancy in species representation and guarantee that biological diversity is taken in consideration in FLR. Moreover, heterogeneous landscape configurations could also improve the provision of ecosystem services at local scale, such as pollination by bees (Ricketts et al. 2008), as opposed to a spatially aggregated approach, as observed in the public lands' allocation of the "Equal Weights" scenario.

Despite the reduced increase in habitat suitability observed for the B75, B50 and EW scenarios, it is important to highlight the benefits of including the likelihood of regeneration as a prioritization criterion. Next to the LRs, the BFC requires native vegetation conservation at hill tops, sloping areas and buffers around riparian areas in private properties, which consist in a protected areas category labeled as Permanent Protection Areas (PPAs), not accounted by our study. In line with Molin et al. (2018), we observed stronger weights of evidence and a prevalence of regeneration close the first 300 meters from the drainage network (Supplementary Figure IV-6). Steeper areas (> 8%) also influenced regeneration (Supplementary Figure IV-4). Favoring such areas contributes to the recovery of PPAs and to increased law compliance. Additionally, the recovery of riparian areas will also contribute to freshwater conservation, prevention of soil erosion, favor animal movement and seed dispersal (Van Looy et al. 2017).

We found that forest restoration in protected areas is a good opportunity for FLR in MT (≈0.1 Mha). Restoration in protected areas involves less costs and higher chances of success due to less intensive land use history and pressure, making it a cost-efficient option for carbon mitigation (Graham et al. 2016). In smaller extent, other land categories could also present feasible options, if law enforcement is increased (undesignated public lands) or cash transfer programs become available (Indigenous Lands). However, the bulk of restoration (≈0.8 Mha) allocated in public lands had "unknown tenure", which may include public undesignated lands, grabbed public lands or non-registered lawful landholdings. Our results highlight the need for improved territorial information and management to clarify the extent of forest restoration potential in public lands (Azevedo-Ramos and Moutinho 2018).

## 4.1. Limitations, implementation and future directions

One important limitation of our study is the assumption of habitat maturation at the early stages of forest recovery, which calls for a careful evaluation of trade-offs between species representation gains. Ideally, the time lag in biodiversity recovery should be considered when allocating restoration targeting species with habitat-demanding traits (Vesk et al. 2008). Thomson et al (2009) tackled this challenge and simulated restoration prioritization considering a schedule that spanned over two centuries. However, a long-term schedule does not apply to our case study, as the BFC requires restoration for LR compensation to take place in a maximum 20 years timeframe impossible.

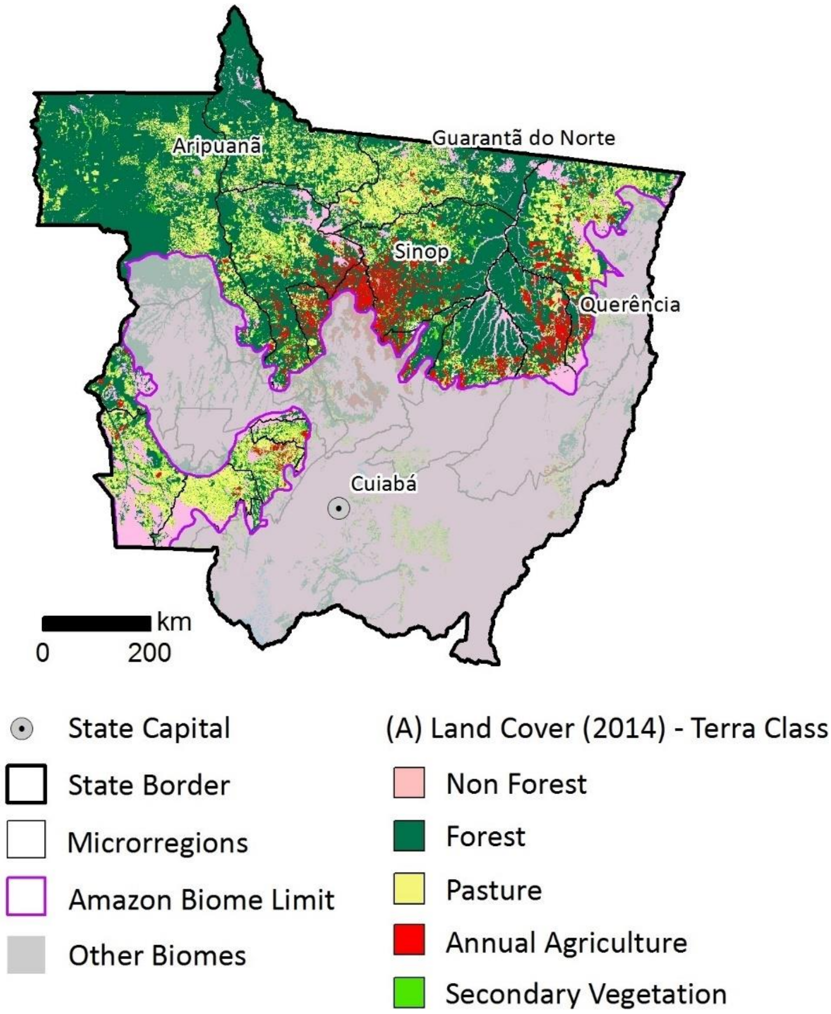
Here, we present forest restoration as a net-cost endeavor. However, forest restoration offers several benefits beyond carbon sequestration that we do not account for, with the potential to reverse restoration from a net-cost to a net-gain status (Brancalion et al. 2012; de Groot et al. 2013). Besides law enforcement and ecosystem services, forest restoration programs could also aim to increase land output producing food, timber and non-timber forest goods, which is allowed by the BFC in LRs under sustainable management (Brasil 2012). Monetization of a wider range of services (e.g., water conservation and erosion control) and goods could be included in future analyses and would require a deeper understanding of the drivers of such activities and economics of multi-target restoration. In the future, prioritization assessments could also be coupled with deforestation and climate change projections (Zwiener et al. 2017) and account for economies of scale in restoration (Strassburg et al. 2019).

Carbon offsetting costs fall within the range of values found by previous studies (Molin et al. 2018; Nunes et al. 2017). However, values are still higher than opportunity costs of avoided emissions from deforestation, which are a priority given the vast areas of unprotected old growth forests in Brazil (Soares-Filho et al. 2014). Therefore, to make restoration feasible, MT should leverage on the BFC potential to expand forests, especially on priority areas, coordinating the major effort of law enforcement with programs to capture financial resources and bring economic rewards to producers, strengthening restoration supply chains and overcoming logistic obstacles (Brancalion et al. 2012; Brancalion et al. 2017).

## 5. Conclusions

Prioritization assessments at national or sub-national level are important to address the challenges to meet restoration commitments. Our prioritization approach could serve as a useful policy instrument to inform cost-effective FLR planning in MT; it allowed us to track habitat gains for species individually and estimate potential carbon gains across scenarios, contrasting such benefits with the costs of implementing restoration. Our findings highlight the importance of the BFC enforcement to make restoration targets viable but also secure biodiversity gains through FLR. The selection of key properties was found to be strategic for a synergistic combination between FLR, law compliance and forest functions enhancement. However, to make the most out of large-scale restoration in MT, it will be important to create supporting policies, complementary to the BFC, to attract investments and engage key landholders in restoration.

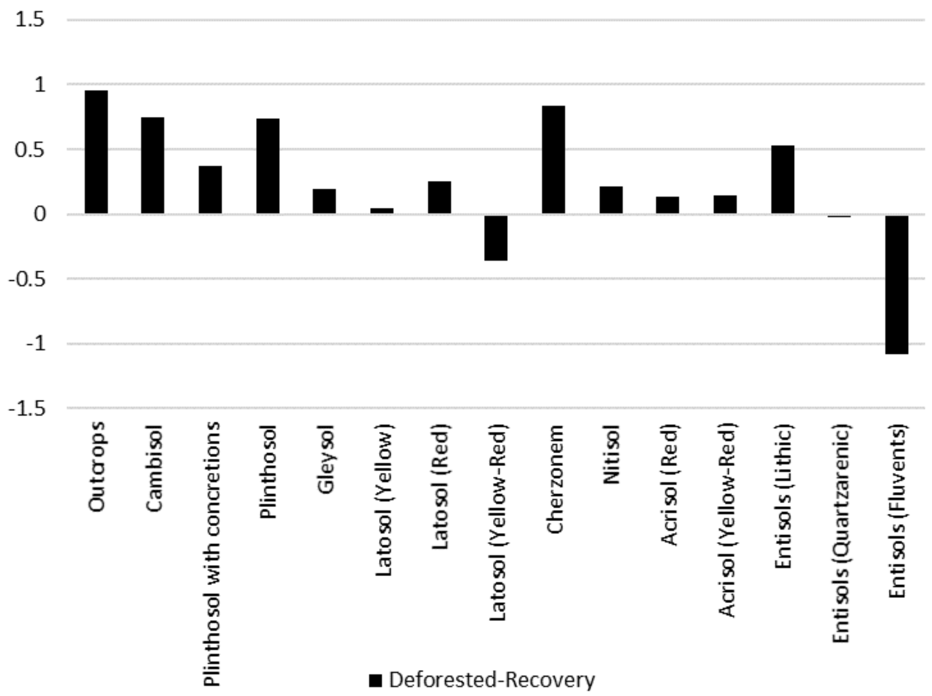
# 6. Supplementary material



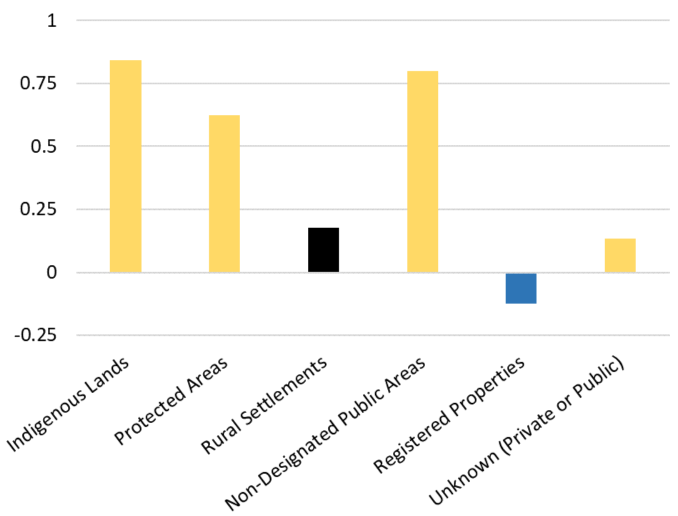

**Supplementary Figure IV-1. Mato Grosso state political limits, study area extent, and land cover categories.** Colored shades highlight the share of MT state contained by the Amazon biome, while greyish shaded areas belong either to the Cerrado or Pantanal biomes. Dark green shade = old-growth (OG) forests, light green shade = secondary (regrowing) vegetation (RF) in areas previously occupied by OG forest, red shade = annual agriculture in areas previously occupied by OG forest; pale yellow shade = grasses established for pasture in areas previously occupied by OG forest; salmon shade = non-forest areas occurring in the Amazon for which land cover change monitoring is not provided by the data source. Source (INPE 2014c).

**Supplementary Text IV-1. Weights of evidence calibration in Dinamica Ego**

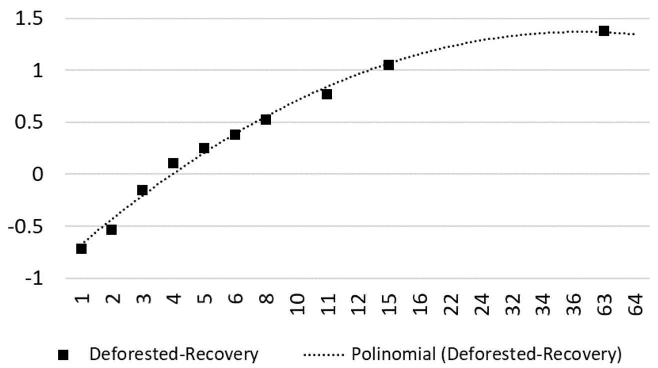
First, we tested the influence of a pre-selected set of variables on the transition from agriculture (i.e., pastureland or cropland) to secondary forest. The WoE requires all variables to be categorical, so we defined ranges for continuous layers (e.g., slope, altitude and distances). Then, we excluded variables with a prevailing inconsistent or null effect on deforestation (-0.5 < WoE < 0.5) (Soares-Filho et al. 2009). Second, we selected all relevant determinants and tested for spatial independency assumption (Soares-Filho et al. 2009) using the Crammer coefficient (V) as a parameter. Our final selection encompassed a set of biophysical (soils, slope or distance to rivers) and socio-economic (distance to deforestation, settlements and infrastructure and level of protection) variables, consistent with the previous literature on spatial determinants of forest regrowth (Molin et al. 2017; Monteiro et al. 2018; Teixeira et al. 2009).



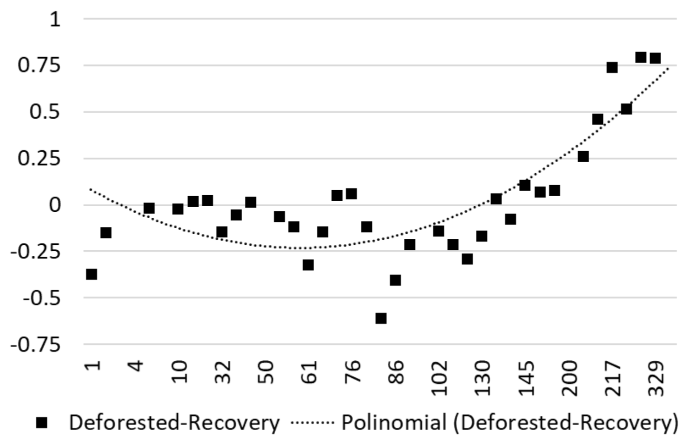

**Supplementary Figure IV-2. Influence (weights of evidence - WoE) of soil classes on deforested-secondary vegetation land cover transition (2012-2014).** Unsuitable soils for agriculture favor land abandonment and subsequent forest recovery (e.g., WoE > 0.5 for outcrops, entisols, cambisol, plinthosol and entisol lithic). Latosol and nitisol known to be fertile or respond well to correctives had negative or negligible WoE. Entisol (fluvent) also showed negative weights, either due to its very small occurrence in MT or for being a poor soil class, leading to arrested forest recovery. Chernozems, on the other hand, are very fertile and showed positive weights. One possible explanation could be that in these lands, forest recovery occurs rapidly after abandonment, even if it lasts a short period.



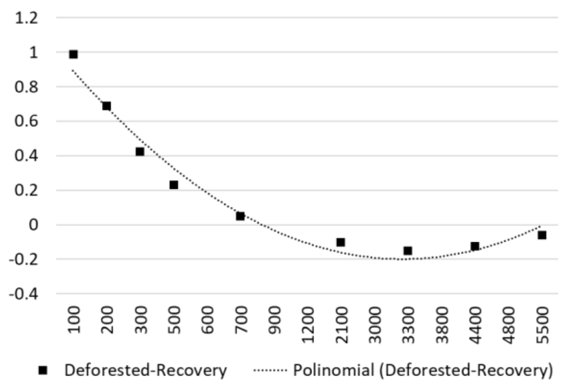

**Supplementary Figure IV-3. Influence (weights of evidence - WoE) of land categories on deforested-secondary vegetation land cover transition (2012-2014).** Indigenous lands, protected areas and non-designated public lands showed positive weights, favoring forest recovery. Rural settlements (+), unknown tenure (+) and registered properties (-) showed either positive or negative, but nearly negligible weights. Likely, this reflects a more intensive agricultural land use in these areas, less prone to abandonment, especially in CAR registered properties.

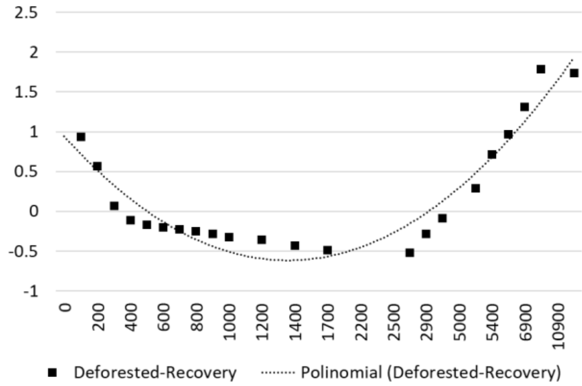



**Supplementary Figure IV-4. Influence (weights of evidence - WoE) of slope (%) on deforested-secondary vegetation land cover transition (2012-2014).** Flat areas showed negative weights, likely due to suitability to mechanized agriculture and intensive land use, less prone to abandonment; on the other hand, higher slopes were more prone to abandonment.

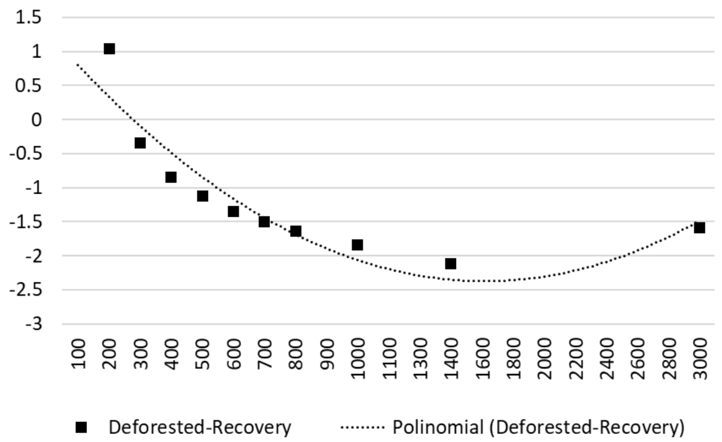



**Supplementary Figure IV-5. Influence (weights of evidence - WoE) of distance (1000 m) paved roads on deforested-secondary vegetation land cover transition (2012-2014).** Weights increased with distance to roads, showing a positive effect on forest regeneration; areas closer to roads are less prone to abandonment and regeneration likely for being more profitable for agriculture due to reduced transportation costs, as opposed to areas far from roads, possibly prohibitive for agriculture and more prone to abandonment following deforestation.

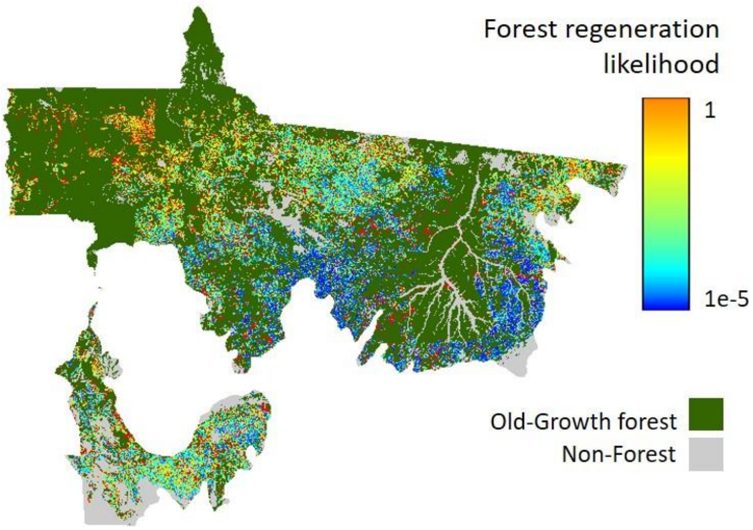


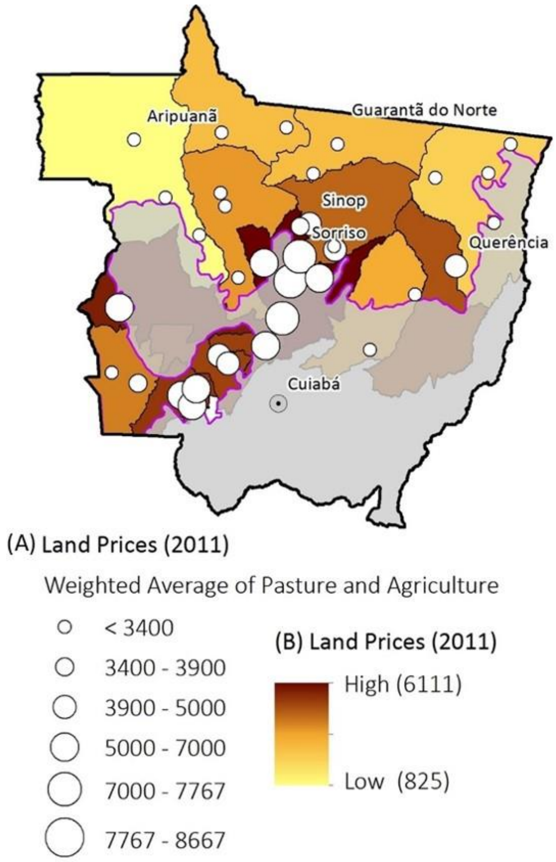

**Supplementary Figure IV-6. Influence (weights of evidence - WoE) of distance (m) to drainage (left) and main rivers (right) on deforested-secondary vegetation land cover transition (2012-2014).** Distance to rivers showed a positive and relevant influence on regeneration until around 400 meters using the drainage network and 200 meters using the main rivers layer after that, weights became close to zero (no-influence). One possible interpretation is that farmers may be allowing regeneration close to rivers, as riparian forests are strict protected by the BFC. Second, distance to main rivers shows a positive effect on regeneration again after 5,000 meters possibly because the distance to water makes land marginal to agriculture and ranching.



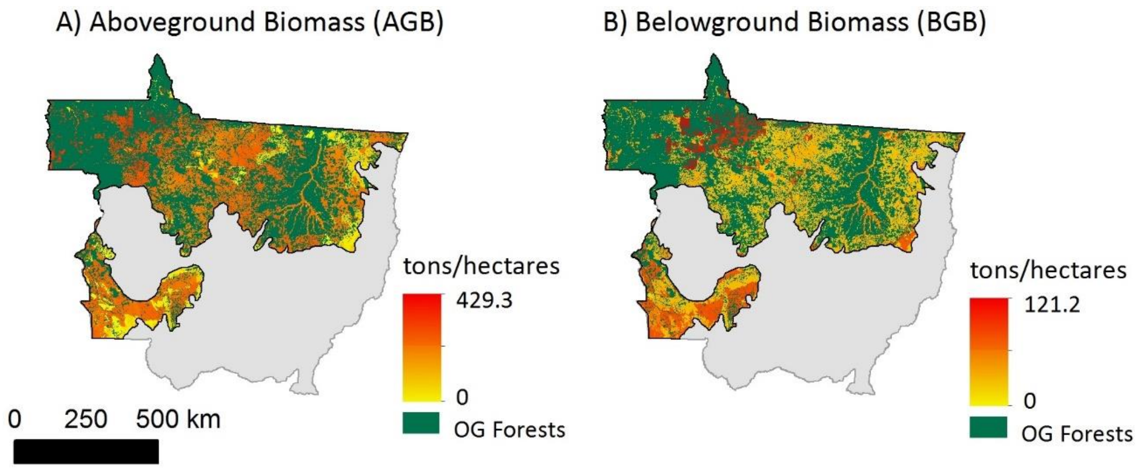

**Supplementary Figure IV-7. Influence (weights of evidence - WoE) of distance (m) to forest edge on deforested-secondary vegetation land cover transition (2012-2014).** Distance to forests had a positive effect on regeneration until 200 meters. As pixels were located further away from edges, showed a decreasingly negative effect on regeneration. This is likely because forests are source of seeds which favors forest regeneration of deforested areas near edges. However, as distance increases this edge effect weakens, which combined with agricultural use repels regeneration.




**Supplementary Figure IV-8. Forest regeneration likelihood.** Spatial distribution of the likelihood (0-1) of clear-cut forest areas, occupied by pasture or croplands, to naturally regenerate following abandonment. Likelihood was estimated based on the identified transitions from agriculturally used lands to regrowing vegetation between 2012 and 2014 as measured by Terra Class (INPE 2014c) and key drivers of regeneration\abandonment. Lowest likelihood values were identified for areas of cropland prevalence (southern edge of the study area, see Supplementary Figure IV-1), and increased towards northwestern MT, to recently cleared areas, proximate to OG forest edges.



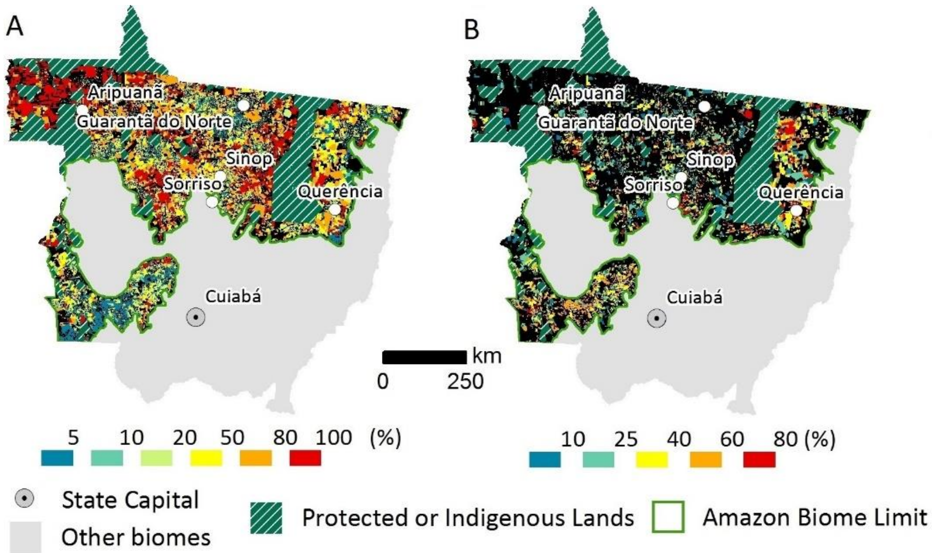

**Supplementary Figure IV-9. Land prices (R$/hectare) in MT.** (A) Punctual depictions show land prices for areas cultivating pasture or annual crops, values varied between R$1,733.0 (pasture, microregion of Aripuanã) and R$8,667.0 (annual crop, microregions of Sinop and Alto Teles Pires). (B) land prices average per microregion, weighted by the pasture or annual cropland cover extent in each microregion. Source: FNP (2012).



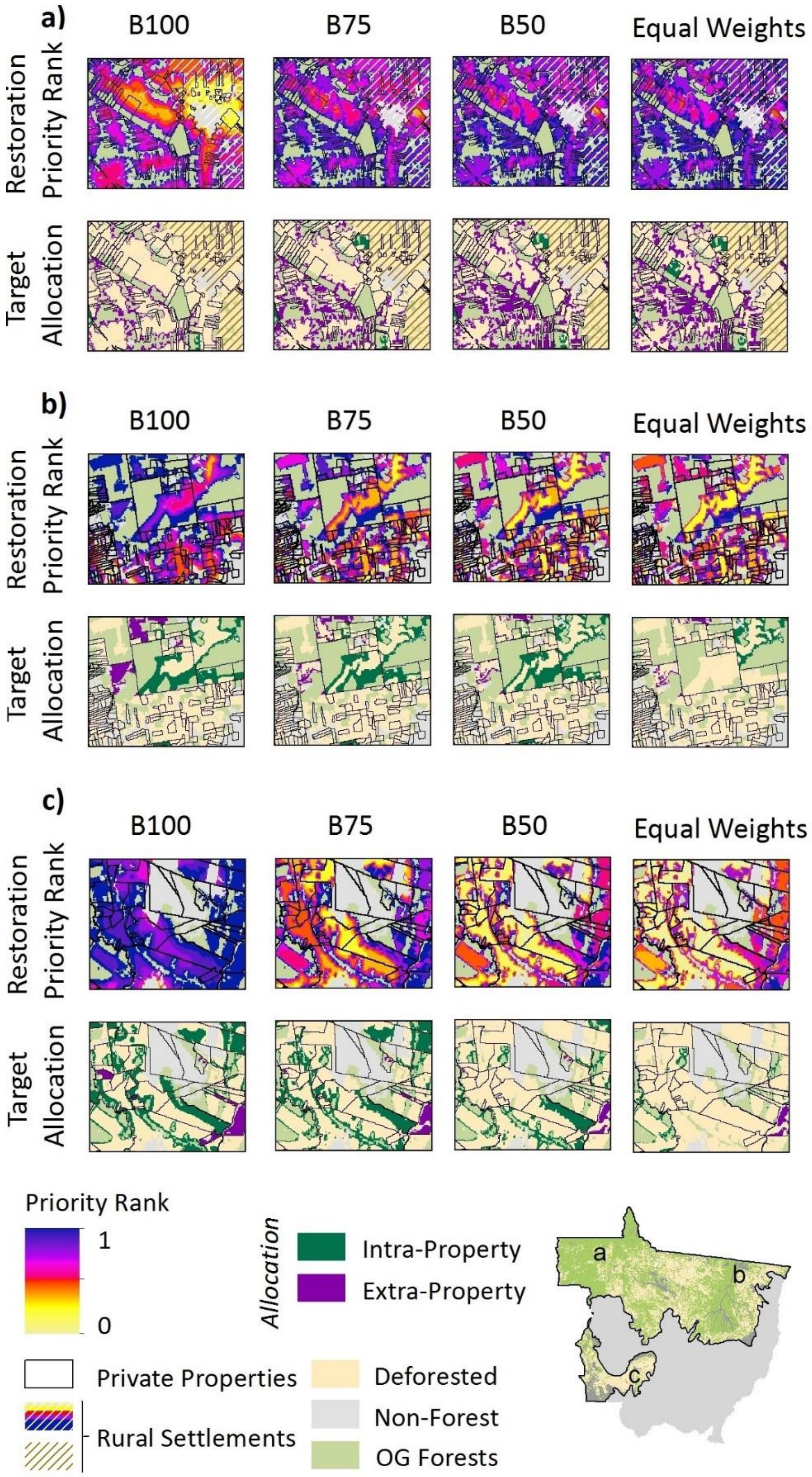

**Supplementary Figure IV-10. (A) Aboveground and (B) belowground biomass reference values for old-growth (OG) forests.** Yellow-Red color scale shows the distribution of reference values of aboveground (A) and belowground (B) biomass, over deforested (2014) areas. Dark-green shade shows OG forests in 2014. Values are measured in tons per hectare. Biomass reference values were used for calculating carbon storage in secondary vegetation in 2014 and the future carbon sequestration potential of areas selected as priority for restoration. Source: Brasil (2016b).

**Supplementary Text IV-2. Calculating the demand for Legal Reserve (LR) deficit restoration**

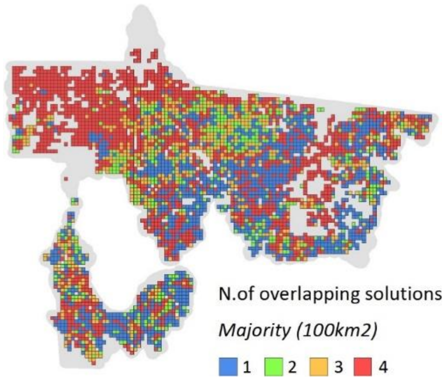
The current version of the BFC specifies that 80% of the forest cover in private lands in the Amazon biome must be protected as LR. For properties with less forest cover the required reparation may range from 0% to the full 80% of the landholding, depending on several factors such as property size, municipality protection level and date of deforestation (Brancalion et al. 2016a; Brasil 2012). To achieve compliance, deficits from deforestation prior to 2008 may be recovered on the property or offset via compensation mechanisms, but on-site forest reestablishment is mandatory for all areas deforested after 2008 (Brasil 2012). For MT, post-2008 deforestation, where revegetation is mandatory, comprehended ≈ 0.4 Mha (INPE 2014b) and was not considered by this prioritization study because their location is already known. To calculate deficits, we used land cover maps (INPE 2014c) and data from a georeferenced land registry of property boundaries (i.e., Environmental Rural Registry - CAR, Portuguese acronym) (Brasil 2012; SICAR 2017). We downloaded 111,487 property polygons from MT and excluded features presenting inconsistencies (e.g., duplicated or overlapping features). Properties smaller than 1.0 ha or entirely covered by non-forest natural areas were also excluded. Our final dataset consisted of 48,160 rural properties. Supplementary Figure IV-11a shows the distribution of forest percentage inside private properties and Supplementary Figure IV-11b shows the distribution of forest deficits across properties in MT. For more details on the BFC forest deficit calculations and the property dataset please refer to Gollnow et al. (2018) and Hissa et al. (2019).



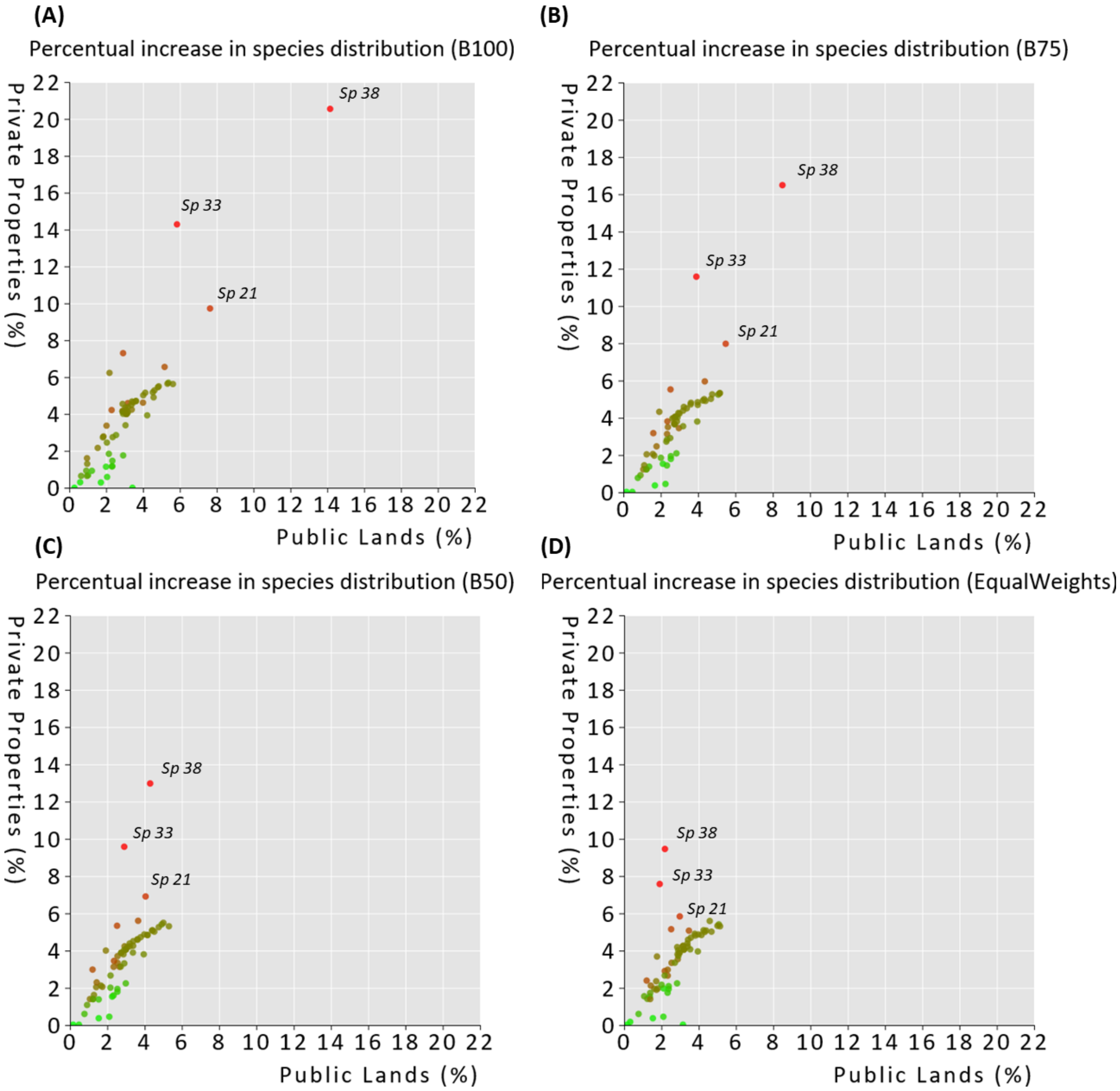

**Supplementary Figure IV-11. (A) Forest cover extent as percentage (%) of the private property area; (B) Private properties classified according to LR deficit extent (%) (SICAR 2017).**



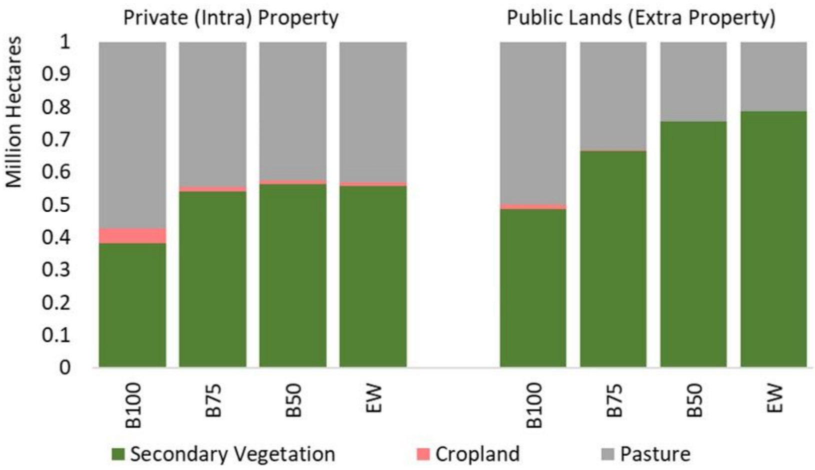

**Supplementary Figure IV-12. Inset panels (a, b and c) of restoration priority and allocation maps across scenarios.** The first line in each of the panels shows the priority ranks and the second line the output of the restoration target allocation at the local level, differentiating between intra (dark-green shade) and extra-property (dark-purple shade) allocation.



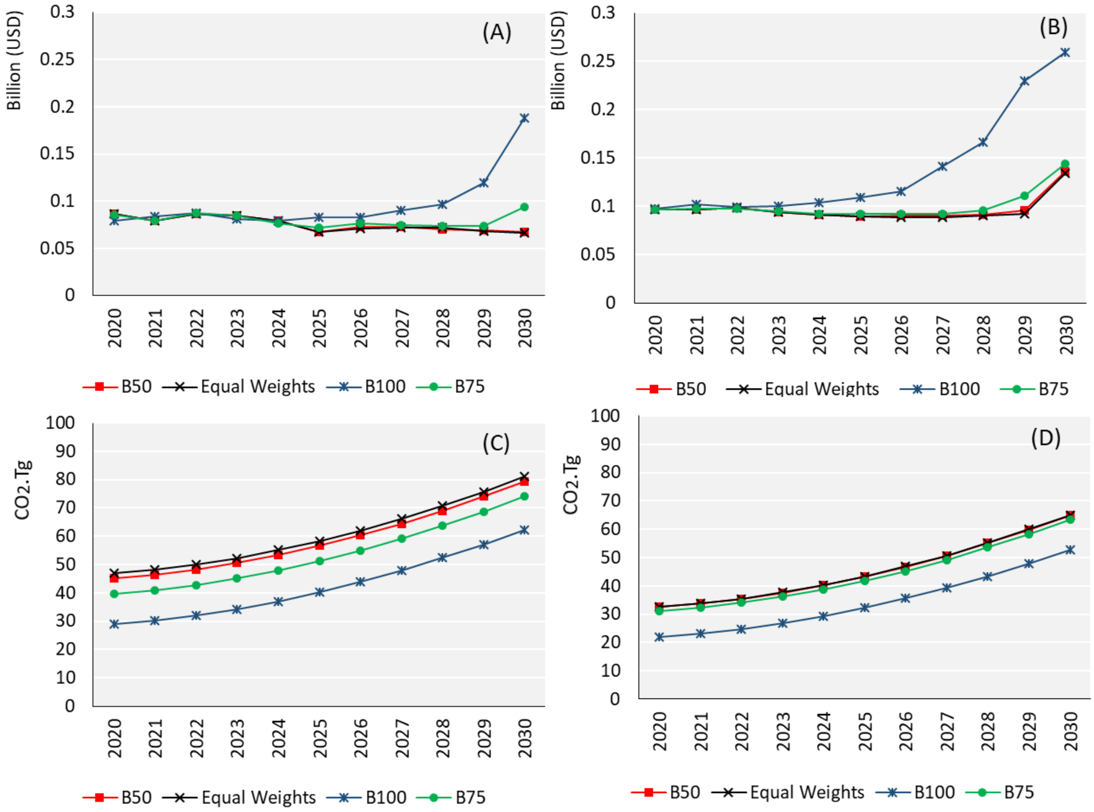

**Supplementary Figure IV-13. Spatial distribution of the overlap in allocated restoration across scenarios.** A maximum of four (all scenarios) and a minimum zero allocations are possible (grey shade). Overlap between solutions was calculated for using a grid of cells of 10x10 km, using a majority filter.




**Supplementary Figure IV-14. Increase (%) in species distributions in private properties (intra-property) versus public lands (extra-property) for each scenario.** Color scale indicates the proportion of OG forest in 2014 covering each species geographic range. Color scale: Bright green = maximum proportion; Bright red = minimum proportion. Sp23= *Dasyprocta azarae*; Sp35= *Leopardus guttulus*; Sp41= *Mazama gouazoubira*;




**Supplementary Figure IV-15. Land cover classes (2014) of areas selected for forest restoration allocation.** The figure shows the land cover classes of selected areas for the allocation of forest restoration targets for each scenario. In the left, land cover classes of selected areas in private lands; in the right, land cover classes of selected areas in public lands. Allocation to secondary vegetation areas (dark green shade) increased from B100 to EW and from private properties to public lands. Pasture areas (grey shade) were the second most important land cover, followed by croplands (salmon shade), to which only a small fraction of restoration was allocated.




**Supplementary Figure IV-16. Forest restoration costs in private (A) and public (B) lands and accumulative carbon stocks annually in private (C) and public (D) lands selected for forest restoration.** Values depicted are annual estimates between 2020 and 2030. Costs estimates (A, B) include total costs (i.e., the sum between opportunity costs and direct restoration costs), represented in USD billions. Accumulated carbon stocks include the sum between carbon previously stored in selected secondary vegetation areas and sequestration between 2019 and 2030.

**Supplementary Table IV-1. Dimensions and indicators for restoration prioritization and trade-offs calculation.**

| Dimension | Indicator | Information Used | Details | Original Resolution | Source |
|---|---|---|---|---|---|
| **Forest Functions Enhancement** | [1] Potential Carbon accumulation by regrowing (2019-2030) forests + Carbon stored in ongoing regeneration (by 2019) | Reference map - original aboveground (AGB) and belowground biomass (BGB) of OG forests | AGB 0-429.3 (mean 207.0 SD 60.8)<br>BGB 0-121.2 (mean 28.64 SD 19.31) | 1000x1000 m | Brasil (2016b) |
| | | Carbon stocks in SG forests selected as priority | Secondary vegetation age (1-11 years) by 2014 | 30x30 m | Terra Class - INPE (2014c) |
| | | Annual Biomass Recovery | 1.2% year | - | Lennox et al. (2018) |
| | [2] Habitat Enhancement for Multiple Species | Geographical Range of non-flying mammals | 65 species | Vector layer, (1:1,000,000 Digital Chart of the World base layer) | IUCN (2018) |
| | | Habitat condition – Reclassified land cover map according to degradation level | OG forest (1.00)<br>SG forest (0.15-0.30)<br>Pasture (0.10)<br>Crop (0.01) | 30x30 m | TerraClass - INPE (2014c) |
| | | Functional Connectivity - Buffers and corridors around high quality patches are preferred according to home range of non-flying mammals | 62 species [1-14,000 hectares] | 100x100m | Table S2<br>TerraClass (INPE 2014c) |
| **Feasibility** | [3] Opportunity Costs | Average land prices for agriculture and pasture (2011). Extrapolated from points to microregions. | 822.7-6,111.0<br>(mean 2,703.6 SD 1,116.3) | - | FNP (2012)<br>TerraClass - (INPE 2014c) |
| | [4] Forest regeneration Likelihood | Simulated forest regeneration likelihood map, based on the 2012-2014 deforested>forested transition and correlated drivers. | * Land use and cover (Calibration 2012,2014)<br>* Selected Drivers<br>- Slope, Soil, Land tenure and Distance: roads, rivers and forest edges | Several, resampled to 100x100m | SICAR (2017), CSR (2017), DNIT (2010), ANA (2010), INPE (2014c), SRTM (2003) |
| **Candidate areas for restoration** | [5] Intra-property allocation | Share of the private property (0-80%) area below the BFC LR requirement | Total LR deficit (3.4 Mha) | Vector layer, 1:50,000 | Hissa et al. (2019), SICAR (2017) |
| | [6] Extra-property allocation | Protected or Indigenous areas, non-designated public lands or unknown tenure | Total eligible area (4.4 Mha) | Several, resampled to 100x100m | CSR (2017) |

**Supplementary Table IV-2. Selected species home range.** List of (62) selected species of terrestrial mammals occurring in Amazonian MT with available home range information.

| N. | Species | Home Range | Reference | N. | Species | Home Range | Reference |
|---|---|---|---|---|---|---|---|
| **1** | *Alouatta belzebul* | 17 | IUCN (2018) | **32** | *Lagothrix cana* | 1,023 | Peres (1996) |
| **2** | *Alouatta caraya* | 19 | Ludwig (2006) | **33** | *Leopardus guttulus* | 2,500 | Kasper et al. (2016) |
| **3** | *Alouatta discolor* | 63 | IUCN (2018) | **34** | *Leopardus pardalis* | 2,263 | Oliveira (2011) |
| **4** | *Alouatta nigerrima* | 45 | IUCN (2018) | **35** | *Leopardus tigrinus* | 1,700 | Myers et al (2018) |
| **5** | *Alouatta seniculus ssp. puruensis* | 10 | Calouro et al. (2015) | **36** | *Leopardus wiedii* | 2,000 | Oliveira (2011) |
| **6** | *Aotus azarae* | 12 | IUCN (2018) | **37** | *Mazama americana* | 52 | Maffei and Taber (2003) |
| **7** | *Aotus nigriceps* | 14 | IUCN (2018) | **38** | *Mazama gouazoubira* | 52 | Maffei and Taber (2003) |
| **8** | *Artibeus lituratus* | 3 | Myers et al (2018) | **39** | *Mazama nemorivaga* | 52 | Maffei and Taber (2003) |
| **9** | *Atelocynus microtis* | 10,000 | Michalski (2010) | **40** | *Mico emiliae* | 40 | IUCN (2018) |
| **10** | *Blastocerus dichotomus* | 6,400 | Perin (2010) | **41** | *Mico intermedius* | 40 | IUCN (2018) |
| **11** | *Bradypus variegatus* | 1 | Castro (2017) | **42** | *Mico melanurus* | 40 | IUCN (2018) |
| **12** | *Caluromys philander* | 3 | Myers et al (2018) | **43** | *Mico nigriceps* | 40 | IUCN (2018) |
| **13** | *Cebus albifrons* | 150 | Ravetta and Muniz (2015) | **44** | *Myrmecophaga tridactyla* | 1,080 | Miranda (2004) |
| **14** | *Cerdocyon thous* | 900 | Myers et al (2018) | **45** | *Nasua nasua* | 588 | Beisiegel and Mantovani (2006) |
| **15** | *Chiropotes albinasus* | 350 | Pinto et al. (2015) | **46** | *Panthera onca* | 13,900 | Cullen (2006) |
| **16** | *Chiropotes utahickae* | 100 | Alonso and Carvalho (2015) | **47** | *Pecari tajacu* | 168 | Jones et al. (2009) |
| **17** | *Chrysocyon brachyurus* | 115 | de Paula et al. (2015) | **48** | *Plecturocebus bernhardi* | 7 | Miranda et al. (2015) |
| **18** | *Coendou prehensilis* | 14 | Jones et al. (2009) | **49** | *Plecturocebus cinerascens* | 7 | Miranda et al. (2015) |
| **19** | *Cuniculus paca* | 4 | Myers et al (2018) | **50** | *Plecturocebus moloch* | 7 | Miranda et al. (2015) |
| **20** | *Cyclopes didactylus* | 4 | Jones et al. (2009) | **51** | *Potos flavus* | 5,300 | Reis et al. (2006 ) |
| **21** | *Dasyprocta azarae* | 8 | Silvius and Fragoso (2003) | **52** | *Procyon cancrivorus* | 695 | Bianchi (2009) |
| **22** | *Dasyprocta leporina* | 8 | Silvius and Fragoso (2003) | **53** | *Puma concolor* | 12,900 | Jones et al. (2009) |
| **23** | *Dasypus kappleri* | 1 | Silva and Barros Henriques (2009) | **54** | *Saimiri sciureus* | 68 | Jones et al. (2009) |
| **24** | *Dasypus novemcinctus* | 21 | Myers et al (2018) | **55** | *Saimiri ustus* | 250 | Jones et al. (2009) |
| **25** | *Dasypus septemcinctus* | 2 | Faria-Corrêa et al. (2015) | **56** | *Sapajus apella* | 350 | Alves et al. (2015) |
| **26** | *Didelphis albiventris* | 2 | Sanches et al. (2012) | **57** | *Sapajus libidinosus* | 150 | Fialho (2015) |
| **27** | *Didelphis marsupialis* | 165 | Myers et al (2018) | **58** | *Speothos venaticus* | 14,000 | Lima et al. (2012) |
| **28** | *Eira barbara* | 2,400 | Emmons and Feer (1997) | **59** | *Sylvilagus brasiliensis* | 1 | Myers et al (2018) |
| **29** | *Galictis vittata* | 415 | Myers et al (2018) | **60** | *Tamandua tetradactyla* | 380 | Ohana et al. (2015) |
| **30** | *Herpailurus yagouaroundi* | 10,000 | IUCN (2018) | **61** | *Tapirus terrestris* | 200 | Jones et al. (2009) |
| **31** | *Hydrochoerus hydrochaeris* | 7 | Jones et al. (2009) | **62** | *Tayassu pecari* | 4,170 | Jones et al. (2009) |

**Supplementary Table IV-3. Home Range, Old Growth (OG) forest cover and original species distribution proportion.**

| Species* | Home Range (hectare) | OG Forests Area (Mha) | Species Occurrence Area (Mha) | OG forests (%) | Original SD prop. | Species* | Home Range (hectare) | OG Forests Area (Mha) | Species Occurrence Area (Mha) | OG forests (%) | Original SD prop. |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Sp1** | 17 | 1.42 | 2.40 | 59.26 | 0.89 | **Sp32** | 1,023 | 7.69 | 9.48 | 81.13 | 0.91 |
| **Sp2** | 18.75 | 7.80 | 13.15 | 59.32 | 0.90 | **Sp33** | 2,500 | 0.64 | 2.62 | 24.60 | 0.43 |
| **Sp3** | 63 | 3.09 | 4.29 | 71.99 | 0.90 | **Sp34** | 2,263 | 26.17 | 42.25 | 61.94 | 0.67 |
| **Sp4** | 45 | 0.20 | 0.20 | 98.30 | 0.99 | **Sp35** | 1,700 | 25.53 | 39.64 | 64.40 | 0.71 |
| **Sp5** | 10 | 7.74 | 9.16 | 84.48 | 0.95 | **Sp36** | 2,000 | 26.17 | 42.25 | 61.94 | 0.67 |
| **Sp6** | 12 | 18.54 | 32.82 | 56.48 | 0.87 | **Sp37** | 52.2 | 26.17 | 42.25 | 61.94 | 0.81 |
| **Sp7** | 14 | 7.63 | 9.43 | 80.93 | 0.95 | **Sp38** | 52.2 | 0.44 | 1.84 | 24.00 | 0.61 |
| **Sp8** | 2.5 | 26.17 | 39.31 | 66.58 | 0.88 | **Sp39** | 52.2 | 25.73 | 40.43 | 63.65 | 0.82 |
| **Sp9** | 10,000 | 3.40 | 3.69 | 92.19 | 0.94 | **Sp40** | 40 | 4.42 | 7.62 | 58.01 | 0.87 |
| **Sp10** | 6,400 | 22.20 | 37.90 | 58.57 | 0.66 | **Sp41** | 40 | 3.30 | 3.57 | 92.66 | 0.97 |
| **Sp11** | 1.29 | 26.15 | 42.22 | 61.93 | 0.89 | **Sp42** | 40 | 4.32 | 9.11 | 47.42 | 0.81 |
| **Sp12** | 3 | 16.10 | 29.41 | 54.76 | 0.89 | **Sp43** | 40 | 0.00 | 0.00 | 94.49 | 0.97 |
| **Sp13** | 150 | 3.95 | 4.50 | 87.84 | 0.95 | **Sp44** | 1,080 | 26.17 | 42.25 | 61.94 | 0.70 |
| **Sp14** | 900 | 5.45 | 11.65 | 46.80 | 0.73 | **Sp45** | 588 | 26.17 | 42.25 | 61.94 | 0.72 |
| **Sp15** | 350 | 15.92 | 22.69 | 70.15 | 0.86 | **Sp46** | 13,900 | 20.21 | 25.61 | 78.94 | 0.82 |
| **Sp16** | 100 | 1.07 | 2.46 | 43.40 | 0.76 | **Sp47** | 168 | 26.17 | 42.25 | 61.94 | 0.77 |
| **Sp17** | 115 | 6.39 | 13.62 | 46.94 | 0.79 | **Sp48** | 6.9 | 2.06 | 2.31 | 89.10 | 0.98 |
| **Sp18** | 14 | 26.17 | 42.25 | 61.94 | 0.84 | **Sp49** | 6.9 | 5.39 | 6.82 | 78.93 | 0.95 |
| **Sp19** | 4 | 26.17 | 42.25 | 61.94 | 0.87 | **Sp50** | 6.9 | 11.66 | 19.54 | 59.65 | 0.90 |
| **Sp20** | 4 | 10.16 | 12.32 | 82.44 | 0.96 | **Sp51** | 5,300 | 26.17 | 42.25 | 61.94 | 0.64 |
| **Sp21** | 8 | 2.29 | 5.95 | 38.46 | 0.80 | **Sp52** | 695 | 26.17 | 42.25 | 61.94 | 0.72 |
| **Sp22** | 8 | 8.25 | 13.40 | 61.59 | 0.91 | **Sp53** | 12,900 | 26.17 | 42.25 | 61.94 | 0.63 |
| **Sp23** | 1.2 | 20.32 | 31.63 | 64.24 | 0.92 | **Sp54** | 68 | 26.17 | 42.25 | 61.94 | 0.86 |
| **Sp24** | 21 | 26.17 | 42.25 | 61.94 | 0.83 | **Sp55** | 250 | 8.40 | 11.08 | 75.83 | 0.91 |
| **Sp25** | 1.6 | 14.60 | 27.20 | 53.68 | 0.90 | **Sp56** | 350 | 14.52 | 20.76 | 69.97 | 0.87 |
| **Sp26** | 2.33 | 7.75 | 14.73 | 52.63 | 0.90 | **Sp57** | 150 | 11.65 | 21.49 | 54.19 | 0.83 |
| **Sp27** | 165 | 10.91 | 16.38 | 66.59 | 0.88 | **Sp58** | 14,000 | 26.17 | 42.25 | 61.94 | 0.61 |
| **Sp28** | 2,400 | 26.17 | 42.25 | 61.94 | 0.67 | **Sp59** | 1 | 26.17 | 42.25 | 61.94 | 0.89 |
| **Sp29** | 415 | 25.48 | 39.49 | 64.51 | 0.68 | **Sp60** | 380 | 26.17 | 42.25 | 61.94 | 0.74 |
| **Sp30** | 10,000 | 26.17 | 42.25 | 61.94 | 0.63 | **Sp61** | 200 | 26.17 | 42.24 | 61.95 | 0.77 |
| **Sp31** | 7 | 26.17 | 42.25 | | 61.94 | **Sp62** | 4,170 | 26.17 | 42.25 | 61.94 | 0.65 |

Home Range (hectare) = species home range in hectares; OG Forests Area (Mha) = extent of old-growth (OG) forest within the geographical range of the species (in millions of hectares - Mha); Species Occurrence Area (Mha) = Species geographic range area in Mha; OG forests (%) = Percentage of the species geographical range covered by OG forests. Original SD prop. = Species distribution (SD) proportion of remaining suitable habitat prior to the restoration target allocation, calculated using Zonation. Data Sources (INPE 2014b; IUCN 2018).

**Supplementary Table IV-4. Direct restoration costs.** Direct restoration costs the restoration approaches associated with each interval of the likelihood of forest regeneration map. Values were obtained from the Restoration manual for the *Teles Pires* and *Alto Juruena* regions, MT (Timotheo et al. 2016).

| Restoration Strategy | Associated Costs (R$/ha) | Associated Likelihood (%) |
|---|---|---|
| Total planting | 19,505.46 | < 25 |
| Partial planting | 7,022.67 | 25-50 |
| Fencing with maintenance | 4,675.00 | 50-75 |
| Maintenance (no fencing) * | 2,875.00 | |
| Fencing with no maintenance | 2,800.00 | > 75 |
| No fencing no maintenance * | 0.00 | |

* No fencing costs were added for restoration taking place in Indigenous Lands or Protected Areas

**Supplementary Table IV-5. Overlap between selected areas for restoration allocation according to each scenario.** Areas below the dashed line refer to the overlap between solutions allocated inside private properties (intra-property) (average overlap of 69.4% and SD 16.8%); Areas above the dashed line refer to the overlap between solutions allocated in public lands, outside (extra-property) private properties (average overlap of 75.3% and SD 12.0%).

| | | **Public Lands (Extra-Property)** | | | |
|---|---|---|---|---|---|
| | | **Equal Weights** | **B100** | **B50** | **B75** |
| **Private (Intra) Property** | **Equal Weights** | | 561,042 *(56.1%)* | 912,843 *(91.3%)* | 775,082 *(77.5%)* |
| | **B100** | 460,369 *(46.0%)* | | 643,581 *(64.4%)* | 766,743 *(76.7%)* |
| | **B50** | 902,733 *(90.3%)* | 527,343 *(52.7%)* | | 862,424 *(86.2%)* |
| | **B75** | 782,029 *(78.2%)* | 623,069 *(62.2%)* | 869,203 *(86.9%)* | |

**Supplementary Table IV-6. Increment in species distribution following the restoration target allocation (scenario B100). See table footnote for column descriptions.**

| Species | Original SD prop. | SD prop. - B100 S | SD prop. - B100 S/EP | SD prop. - B100 S/IP | (%) Δ DS - B100 S | (%) Δ DS - B100 S/EP | (%) Δ DS - B100 S/IP | Species | Original SD prop | SD prop. - B100 S | SD prop. - B100 S/EP | SD prop. - B100 S/IP | (%) Δ DS - B100 S | (%) Δ DS - B100 S/EP | (%) Δ DS - B100 S/IP |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Sp1** | 0.89 | 0.96 | 0.91 | 0.95 | 8.10 | 2.18 | 6.23 | **Sp32** | 0.91 | 0.93 | 0.92 | 0.91 | 2.15 | 1.23 | 0.92 |
| **Sp2** | 0.90 | 0.94 | 0.91 | 0.92 | 4.64 | 1.86 | 2.79 | **Sp33** | 0.43 | 0.50 | 0.45 | 0.49 | 16.88 | 5.84 | 14.29 |
| **Sp3** | 0.90 | 0.92 | 0.91 | 0.91 | 1.85 | 0.92 | 0.92 | **Sp34** | 0.67 | 0.72 | 0.69 | 0.70 | 7.44 | 2.89 | 4.13 |
| **Sp4** | 0.99 | 0.99 | 0.99 | 0.99 | 0.28 | 0.28 | 0.00 | **Sp35** | 0.71 | 0.74 | 0.73 | 0.73 | 4.71 | 2.35 | 2.75 |
| **Sp5** | 0.95 | 0.98 | 0.97 | 0.96 | 3.51 | 2.34 | 1.17 | **Sp36** | 0.67 | 0.72 | 0.69 | 0.70 | 7.44 | 2.89 | 4.13 |
| **Sp6** | 0.87 | 0.93 | 0.89 | 0.90 | 7.37 | 2.88 | 4.17 | **Sp37** | 0.81 | 0.88 | 0.84 | 0.85 | 9.28 | 4.12 | 5.15 |
| **Sp7** | 0.95 | 0.98 | 0.97 | 0.96 | 3.51 | 2.34 | 1.46 | **Sp38** | 0.61 | 0.78 | 0.69 | 0.73 | 28.77 | 14.16 | 20.55 |
| **Sp8** | 0.88 | 0.97 | 0.93 | 0.93 | 10.76 | 5.38 | 5.70 | **Sp39** | 0.82 | 0.87 | 0.84 | 0.85 | 6.44 | 3.05 | 3.39 |
| **Sp9** | 0.94 | 0.94 | 0.94 | 0.94 | 0.59 | 0.59 | 0.30 | **Sp40** | 0.87 | 0.89 | 0.88 | 0.88 | 2.56 | 0.96 | 1.60 |
| **Sp10** | 0.66 | 0.70 | 0.68 | 0.68 | 7.20 | 3.39 | 4.24 | **Sp41** | 0.97 | 0.99 | 0.99 | 0.98 | 2.00 | 1.71 | 0.29 |
| **Sp11** | 0.89 | 0.98 | 0.93 | 0.94 | 10.97 | 5.33 | 5.64 | **Sp42** | 0.81 | 0.89 | 0.85 | 0.86 | 10.69 | 5.17 | 6.55 |
| **Sp12** | 0.89 | 0.96 | 0.92 | 0.93 | 7.14 | 3.11 | 4.04 | **Sp43** | 0.97 | 1.00 | 1.00 | 0.97 | 3.43 | 3.43 | 0.00 |
| **Sp13** | 0.95 | 0.97 | 0.97 | 0.95 | 2.35 | 2.05 | 0.59 | **Sp44** | 0.70 | 0.75 | 0.72 | 0.73 | 7.57 | 3.19 | 4.38 |
| **Sp14** | 0.73 | 0.77 | 0.74 | 0.76 | 6.13 | 2.30 | 4.21 | **Sp45** | 0.72 | 0.78 | 0.74 | 0.75 | 8.08 | 3.08 | 4.23 |
| **Sp15** | 0.86 | 0.88 | 0.87 | 0.87 | 1.61 | 0.64 | 0.64 | **Sp46** | 0.82 | 0.83 | 0.83 | 0.82 | 1.70 | 1.02 | 0.68 |
| **Sp16** | 0.76 | 0.83 | 0.78 | 0.82 | 9.49 | 2.92 | 7.30 | **Sp47** | 0.77 | 0.83 | 0.80 | 0.81 | 8.30 | 3.61 | 4.69 |
| **Sp17** | 0.79 | 0.84 | 0.81 | 0.82 | 7.42 | 3.18 | 4.59 | **Sp48** | 0.98 | 1.00 | 0.99 | 0.99 | 2.85 | 1.99 | 1.14 |
| **Sp18** | 0.84 | 0.92 | 0.88 | 0.89 | 9.57 | 4.62 | 5.28 | **Sp49** | 0.95 | 0.99 | 0.98 | 0.97 | 4.39 | 2.92 | 1.75 |
| **Sp19** | 0.87 | 0.96 | 0.91 | 0.91 | 10.26 | 4.81 | 5.45 | **Sp50** | 0.90 | 0.93 | 0.91 | 0.92 | 3.70 | 1.54 | 2.16 |
| **Sp20** | 0.96 | 0.99 | 0.98 | 0.97 | 3.47 | 2.31 | 1.16 | **Sp51** | 0.64 | 0.69 | 0.66 | 0.67 | 7.39 | 3.04 | 4.35 |
| **Sp21** | 0.80 | 0.93 | 0.86 | 0.88 | 15.63 | 7.64 | 9.72 | **Sp52** | 0.72 | 0.78 | 0.74 | 0.75 | 8.11 | 3.47 | 4.63 |
| **Sp22** | 0.91 | 0.95 | 0.93 | 0.94 | 4.26 | 1.82 | 2.74 | **Sp53** | 0.63 | 0.67 | 0.64 | 0.65 | 7.11 | 3.11 | 4.00 |
| **Sp23** | 0.92 | 0.99 | 0.96 | 0.96 | 8.16 | 4.23 | 3.93 | **Sp54** | 0.86 | 0.88 | 0.86 | 0.87 | 2.27 | 0.97 | 1.30 |
| **Sp24** | 0.83 | 0.91 | 0.86 | 0.87 | 9.36 | 4.01 | 5.02 | **Sp55** | 0.91 | 0.94 | 0.92 | 0.92 | 3.68 | 2.15 | 1.84 |
| **Sp25** | 0.90 | 0.96 | 0.93 | 0.94 | 6.46 | 3.08 | 4.00 | **Sp56** | 0.87 | 0.88 | 0.88 | 0.87 | 1.60 | 0.96 | 0.64 |
| **Sp26** | 0.90 | 0.98 | 0.94 | 0.94 | 8.31 | 4.00 | 4.62 | **Sp57** | 0.83 | 0.87 | 0.84 | 0.85 | 5.05 | 2.02 | 3.37 |
| **Sp27** | 0.88 | 0.92 | 0.90 | 0.90 | 5.08 | 2.54 | 2.86 | **Sp58** | 0.61 | 0.66 | 0.63 | 0.64 | 7.27 | 3.18 | 4.09 |
| **Sp28** | 0.67 | 0.72 | 0.69 | 0.70 | 7.44 | 2.89 | 4.55 | **Sp59** | 0.89 | 0.98 | 0.94 | 0.94 | 10.63 | 5.63 | 5.63 |
| **Sp29** | 0.68 | 0.71 | 0.69 | 0.70 | 4.90 | 2.04 | 2.45 | **Sp60** | 0.74 | 0.80 | 0.77 | 0.78 | 8.24 | 3.37 | 4.49 |
| **Sp30** | 0.63 | 0.67 | 0.64 | 0.65 | 7.11 | 3.11 | 4.00 | **Sp61** | 0.77 | 0.83 | 0.79 | 0.80 | 8.33 | 3.62 | 4.71 |
| **Sp31** | 0.86 | 0.95 | 0.90 | 0.91 | 10.36 | 4.85 | 5.50 | **Sp62** | 0.65 | 0.70 | 0.68 | 0.68 | 7.66 | 3.40 | 4.68 |

*Original SD prop.* = Species distribution (SD) proportion of remaining suitable habitat prior to the restoration target allocation, calculated using Zonation; *SD prop. - B100 S* = SD proportion of suitable habitat following the allocation of the B100 full solution (S); *SD prop. - B100 S/EP* SD proportion of suitable habitat following the allocation of the B100 S for public lands (Extra-Property; EP). *SD prop. - B100 S/IP* proportion of suitable habitat following the allocation of the B100 S for private properties (Intra-Property; IP). *(%) Δ DS - B100 S* Increase in SD obtained by the allocation of the full B100 S; *(%) Δ DS - B100 S/EP* Increase in SD obtained by the allocation of the B100 S for public lands (Extra-Property; EP). *(%) Δ DS - B100 S/IP*; Increase in SD obtained by the allocation of the B100 S for private properties (Intra-Property; IP);

**Supplementary Table IV-7. Increment in species distribution following the restoration target allocation (scenario B75). See table footnote for column descriptions.**

| Species | SD prop. HQH - B100 | SD prop. HQH - B75 S | SD prop. HQH - B75 S/EP | SD prop. HQH - B75 S/IP | (%) Δ in HQH - B75 S | (%) Δ in HQH - B75 S/EP | (%) Δ in HQH - B75 S/IP | Species | SD prop. HQH - 100 | SD prop. HQH - B75 S | SD prop. HQH - B75 S/EP | SD prop. HQH - B75 S/IP | (%) Δ in HQH - B75 S | (%) Δ in HQH - B75 S/EP | (%) Δ in HQH - B75 S/IP |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Sp1** | 0.89 | 0.95 | 0.91 | 0.93 | 6.23 | 1.92 | 4.32 | **Sp32** | 0.91 | 0.93 | 0.92 | 0.92 | 3.07 | 1.38 | 1.38 |
| **Sp2** | 0.90 | 0.93 | 0.91 | 0.92 | 4.02 | 1.58 | 2.05 | **Sp33** | 0.43 | 0.49 | 0.44 | 0.48 | 14.29 | 3.91 | 11.58 |
| **Sp3** | 0.90 | 0.92 | 0.91 | 0.91 | 2.46 | 1.24 | 1.24 | **Sp34** | 0.67 | 0.72 | 0.69 | 0.70 | 7.44 | 2.79 | 4.06 |
| **Sp4** | 0.99 | 0.99 | 0.99 | 0.99 | 0.28 | 0.17 | 0.02 | **Sp35** | 0.71 | 0.75 | 0.73 | 0.73 | 5.49 | 2.51 | 2.91 |
| **Sp5** | 0.95 | 0.99 | 0.97 | 0.97 | 4.68 | 2.54 | 1.79 | **Sp36** | 0.67 | 0.72 | 0.69 | 0.70 | 7.44 | 2.79 | 4.06 |
| **Sp6** | 0.87 | 0.93 | 0.89 | 0.90 | 6.73 | 2.71 | 3.70 | **Sp37** | 0.81 | 0.88 | 0.84 | 0.85 | 9.28 | 3.99 | 4.69 |
| **Sp7** | 0.95 | 0.99 | 0.97 | 0.97 | 4.39 | 2.54 | 1.94 | **Sp38** | 0.61 | 0.74 | 0.66 | 0.71 | 22.37 | 8.52 | 16.49 |
| **Sp8** | 0.88 | 0.97 | 0.92 | 0.92 | 10.44 | 5.08 | 5.25 | **Sp39** | 0.82 | 0.88 | 0.85 | 0.85 | 7.12 | 3.20 | 3.55 |
| **Sp9** | 0.94 | 0.94 | 0.94 | 0.94 | 0.59 | 0.48 | 0.02 | **Sp40** | 0.87 | 0.89 | 0.88 | 0.88 | 2.56 | 1.07 | 1.23 |
| **Sp10** | 0.66 | 0.70 | 0.67 | 0.68 | 7.20 | 2.71 | 4.02 | **Sp41** | 0.97 | 0.99 | 0.99 | 0.98 | 2.29 | 1.69 | 0.37 |
| **Sp11** | 0.89 | 0.98 | 0.93 | 0.93 | 10.66 | 5.17 | 5.33 | **Sp42** | 0.81 | 0.88 | 0.84 | 0.85 | 9.66 | 4.36 | 5.96 |
| **Sp12** | 0.89 | 0.95 | 0.92 | 0.93 | 6.52 | 2.39 | 3.50 | **Sp43** | 0.97 | 1.00 | 1.00 | 0.97 | 3.43 | 3.16 | 0.03 |
| **Sp13** | 0.95 | 0.97 | 0.97 | 0.95 | 2.64 | 2.26 | 0.45 | **Sp44** | 0.70 | 0.75 | 0.72 | 0.73 | 7.57 | 3.00 | 4.23 |
| **Sp14** | 0.73 | 0.76 | 0.74 | 0.75 | 4.98 | 1.60 | 3.18 | **Sp45** | 0.72 | 0.78 | 0.75 | 0.75 | 8.08 | 3.20 | 4.39 |
| **Sp15** | 0.86 | 0.89 | 0.87 | 0.87 | 2.57 | 0.92 | 0.92 | **Sp46** | 0.82 | 0.83 | 0.82 | 0.82 | 1.70 | 0.77 | 0.77 |
| **Sp16** | 0.76 | 0.82 | 0.78 | 0.80 | 8.03 | 2.53 | 5.52 | **Sp47** | 0.77 | 0.83 | 0.80 | 0.80 | 8.30 | 3.40 | 4.51 |
| **Sp17** | 0.79 | 0.83 | 0.80 | 0.82 | 6.01 | 2.36 | 3.81 | **Sp48** | 0.98 | 1.00 | 1.00 | 0.99 | 3.70 | 2.12 | 1.53 |
| **Sp18** | 0.84 | 0.92 | 0.88 | 0.88 | 9.57 | 4.40 | 4.91 | **Sp49** | 0.95 | 1.00 | 0.98 | 0.97 | 4.97 | 2.84 | 2.09 |
| **Sp19** | 0.87 | 0.95 | 0.91 | 0.91 | 9.94 | 4.69 | 5.02 | **Sp50** | 0.90 | 0.93 | 0.91 | 0.92 | 3.70 | 1.25 | 2.04 |
| **Sp20** | 0.96 | 1.00 | 0.98 | 0.97 | 4.05 | 2.33 | 1.44 | **Sp51** | 0.64 | 0.69 | 0.66 | 0.66 | 7.39 | 2.63 | 3.97 |
| **Sp21** | 0.80 | 0.90 | 0.84 | 0.86 | 12.50 | 5.48 | 7.98 | **Sp52** | 0.72 | 0.78 | 0.75 | 0.75 | 8.49 | 3.62 | 4.81 |
| **Sp22** | 0.91 | 0.95 | 0.93 | 0.93 | 3.95 | 1.66 | 1.97 | **Sp53** | 0.63 | 0.67 | 0.64 | 0.65 | 7.11 | 2.75 | 3.66 |
| **Sp23** | 0.92 | 0.99 | 0.96 | 0.95 | 8.16 | 3.96 | 3.80 | **Sp54** | 0.86 | 0.88 | 0.87 | 0.87 | 2.60 | 1.12 | 1.45 |
| **Sp24** | 0.83 | 0.91 | 0.86 | 0.87 | 9.36 | 3.98 | 4.84 | **Sp55** | 0.91 | 0.94 | 0.92 | 0.92 | 3.99 | 2.01 | 1.85 |
| **Sp25** | 0.90 | 0.96 | 0.92 | 0.93 | 5.85 | 2.34 | 3.13 | **Sp56** | 0.87 | 0.89 | 0.88 | 0.88 | 2.56 | 1.23 | 1.23 |
| **Sp26** | 0.90 | 0.96 | 0.93 | 0.93 | 6.77 | 2.97 | 3.45 | **Sp57** | 0.83 | 0.86 | 0.84 | 0.85 | 4.04 | 1.78 | 2.47 |
| **Sp27** | 0.88 | 0.92 | 0.90 | 0.90 | 5.08 | 2.34 | 2.83 | **Sp58** | 0.61 | 0.66 | 0.63 | 0.63 | 7.27 | 2.87 | 3.80 |
| **Sp28** | 0.67 | 0.72 | 0.69 | 0.70 | 7.44 | 2.79 | 4.06 | **Sp59** | 0.89 | 0.98 | 0.93 | 0.94 | 10.63 | 5.15 | 5.31 |
| **Sp29** | 0.68 | 0.72 | 0.70 | 0.70 | 5.31 | 2.30 | 2.72 | **Sp60** | 0.74 | 0.80 | 0.77 | 0.78 | 8.24 | 3.24 | 4.58 |
| **Sp30** | 0.63 | 0.67 | 0.64 | 0.65 | 7.11 | 2.75 | 3.66 | **Sp61** | 0.77 | 0.83 | 0.79 | 0.80 | 8.70 | 3.60 | 4.72 |
| **Sp31** | 0.86 | 0.94 | 0.90 | 0.90 | 10.03 | 4.76 | 5.26 | **Sp62** | 0.65 | 0.70 | 0.67 | 0.68 | 7.66 | 2.95 | 4.26 |

*Original SD prop.* = Species distribution (SD) proportion of remaining suitable habitat prior to the restoration target allocation, calculated using Zonation; *SD prop. - B75 S* = SD proportion of suitable habitat following the allocation of the B75 full solution (S); *SD prop. - B75 S/EP* SD proportion of suitable habitat following the allocation of the B75 S for public lands (Extra-Property; EP). *SD prop. - B75 S/IP* proportion of suitable habitat following the allocation of the B75 S for private properties (Intra-Property; IP). *(%) Δ DS - B75 S* Increase in SD obtained by the allocation of the full B75 S; *(%) Δ DS - B75 S/EP* Increase in SD obtained by the allocation of the B75 S for public lands (Extra-Property; EP). *(%) Δ DS - B75 S/IP*; Increase in SD obtained by the allocation of the B75 S for private properties (Intra-Property; IP);

**Supplementary Table IV-8. Increment in species distribution following the restoration target allocation (scenario B50). See table footnote for column descriptions.**

| Species | SD prop. HQH - B100 | SD prop. HQH - B50 S | SD prop. HQH - B50 S/EP | SD prop. HQH - B50 S/IP | (%) Δ in HQH - B50 S | (%) Δ in HQH - B50 S/EP | (%) Δ in HQH - B50 S/IP | Species | SD prop. HQH - 100 | SD prop. HQH - B50 S | SD prop. HQH - B50 S/EP | SD prop. HQH - B50 S/IP | (%) Δ in HQH - B50 S | (%) Δ in HQH - B50 S/EP | (%) Δ in HQH - B50 S/IP |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Sp1** | 0.89 | 0.94 | 0.91 | 0.93 | 5.92 | 1.92 | 4.00 | **Sp32** | 0.91 | 0.93 | 0.92 | 0.92 | 3.07 | 1.54 | 1.38 |
| **Sp2** | 0.90 | 0.93 | 0.91 | 0.92 | 4.02 | 1.74 | 2.05 | **Sp33** | 0.43 | 0.48 | 0.44 | 0.47 | 11.69 | 2.91 | 9.58 |
| **Sp3** | 0.90 | 0.93 | 0.91 | 0.92 | 2.77 | 1.24 | 1.39 | **Sp34** | 0.67 | 0.73 | 0.69 | 0.70 | 7.85 | 3.00 | 4.06 |
| **Sp4** | 0.99 | 0.99 | 0.99 | 0.99 | 0.28 | 0.17 | 0.02 | **Sp35** | 0.71 | 0.75 | 0.73 | 0.73 | 6.27 | 2.91 | 3.32 |
| **Sp5** | 0.95 | 0.99 | 0.97 | 0.97 | 4.68 | 2.54 | 1.79 | **Sp36** | 0.67 | 0.73 | 0.69 | 0.70 | 7.85 | 3.00 | 4.06 |
| **Sp6** | 0.87 | 0.93 | 0.89 | 0.90 | 6.73 | 2.55 | 3.70 | **Sp37** | 0.81 | 0.88 | 0.84 | 0.85 | 9.28 | 3.99 | 4.87 |
| **Sp7** | 0.95 | 0.99 | 0.97 | 0.97 | 4.68 | 2.54 | 1.94 | **Sp38** | 0.61 | 0.71 | 0.63 | 0.69 | 16.44 | 4.30 | 12.97 |
| **Sp8** | 0.88 | 0.97 | 0.92 | 0.93 | 10.44 | 4.92 | 5.41 | **Sp39** | 0.82 | 0.88 | 0.85 | 0.85 | 7.80 | 3.37 | 3.90 |
| **Sp9** | 0.94 | 0.94 | 0.94 | 0.94 | 0.59 | 0.48 | 0.02 | **Sp40** | 0.87 | 0.89 | 0.88 | 0.88 | 2.56 | 1.07 | 1.40 |
| **Sp10** | 0.66 | 0.70 | 0.67 | 0.68 | 7.20 | 2.93 | 4.23 | **Sp41** | 0.97 | 0.99 | 0.99 | 0.98 | 2.29 | 1.54 | 0.37 |
| **Sp11** | 0.89 | 0.98 | 0.93 | 0.93 | 10.34 | 5.01 | 5.49 | **Sp42** | 0.81 | 0.88 | 0.83 | 0.85 | 8.97 | 3.65 | 5.60 |
| **Sp12** | 0.89 | 0.95 | 0.92 | 0.92 | 6.52 | 2.55 | 3.34 | **Sp43** | 0.97 | 1.00 | 1.00 | 0.97 | 3.43 | 3.16 | 0.03 |
| **Sp13** | 0.95 | 0.97 | 0.97 | 0.95 | 2.64 | 2.11 | 0.45 | **Sp44** | 0.70 | 0.75 | 0.72 | 0.73 | 7.57 | 3.21 | 4.23 |
| **Sp14** | 0.73 | 0.76 | 0.73 | 0.75 | 4.60 | 1.21 | 2.98 | **Sp45** | 0.72 | 0.78 | 0.75 | 0.75 | 8.08 | 3.20 | 4.39 |
| **Sp15** | 0.86 | 0.89 | 0.87 | 0.87 | 2.89 | 0.92 | 1.08 | **Sp46** | 0.82 | 0.83 | 0.82 | 0.82 | 1.70 | 0.77 | 0.60 |
| **Sp16** | 0.76 | 0.82 | 0.78 | 0.80 | 8.03 | 2.53 | 5.34 | **Sp47** | 0.77 | 0.84 | 0.80 | 0.80 | 8.66 | 3.40 | 4.51 |
| **Sp17** | 0.79 | 0.83 | 0.80 | 0.81 | 5.65 | 2.36 | 3.45 | **Sp48** | 0.98 | 1.00 | 1.00 | 0.99 | 3.99 | 2.26 | 1.53 |
| **Sp18** | 0.84 | 0.92 | 0.88 | 0.88 | 9.57 | 4.40 | 5.08 | **Sp49** | 0.95 | 1.00 | 0.98 | 0.97 | 5.26 | 2.99 | 2.24 |
| **Sp19** | 0.87 | 0.95 | 0.91 | 0.91 | 9.94 | 4.52 | 5.02 | **Sp50** | 0.90 | 0.94 | 0.91 | 0.92 | 4.01 | 1.41 | 2.04 |
| **Sp20** | 0.96 | 1.00 | 0.98 | 0.98 | 4.34 | 2.33 | 1.59 | **Sp51** | 0.64 | 0.69 | 0.66 | 0.67 | 7.39 | 3.07 | 4.19 |
| **Sp21** | 0.80 | 0.88 | 0.83 | 0.86 | 10.42 | 4.06 | 6.91 | **Sp52** | 0.72 | 0.78 | 0.75 | 0.75 | 8.49 | 3.62 | 4.61 |
| **Sp22** | 0.91 | 0.95 | 0.93 | 0.93 | 3.95 | 1.66 | 2.12 | **Sp53** | 0.63 | 0.67 | 0.64 | 0.65 | 7.11 | 2.75 | 3.89 |
| **Sp23** | 0.92 | 0.99 | 0.96 | 0.95 | 8.16 | 3.96 | 3.80 | **Sp54** | 0.86 | 0.88 | 0.87 | 0.87 | 3.25 | 1.28 | 1.62 |
| **Sp24** | 0.83 | 0.91 | 0.87 | 0.87 | 9.36 | 4.15 | 4.84 | **Sp55** | 0.91 | 0.94 | 0.93 | 0.92 | 3.99 | 2.17 | 2.01 |
| **Sp25** | 0.90 | 0.96 | 0.92 | 0.93 | 5.85 | 2.34 | 3.13 | **Sp56** | 0.87 | 0.89 | 0.88 | 0.88 | 2.88 | 1.23 | 1.40 |
| **Sp26** | 0.90 | 0.96 | 0.93 | 0.93 | 5.85 | 2.66 | 3.13 | **Sp57** | 0.83 | 0.86 | 0.84 | 0.84 | 3.70 | 1.43 | 2.30 |
| **Sp27** | 0.88 | 0.92 | 0.89 | 0.90 | 4.76 | 2.17 | 2.66 | **Sp58** | 0.61 | 0.66 | 0.63 | 0.63 | 7.27 | 2.87 | 3.80 |
| **Sp28** | 0.67 | 0.73 | 0.69 | 0.70 | 7.85 | 3.00 | 4.06 | **Sp59** | 0.89 | 0.98 | 0.94 | 0.94 | 10.63 | 5.31 | 5.31 |
| **Sp29** | 0.68 | 0.72 | 0.70 | 0.70 | 6.12 | 2.72 | 3.14 | **Sp60** | 0.74 | 0.80 | 0.77 | 0.78 | 8.24 | 3.62 | 4.58 |
| **Sp30** | 0.63 | 0.67 | 0.64 | 0.65 | 7.11 | 2.75 | 3.89 | **Sp61** | 0.77 | 0.83 | 0.80 | 0.80 | 8.70 | 3.79 | 4.72 |
| **Sp31** | 0.86 | 0.94 | 0.90 | 0.90 | 10.03 | 4.76 | 5.26 | **Sp62** | 0.65 | 0.70 | 0.67 | 0.68 | 7.66 | 3.39 | 4.26 |

*Original SD prop.* = Species distribution (SD) proportion of remaining suitable habitat prior to the restoration target allocation, calculated using Zonation; *SD prop. - B50 S* = SD proportion of suitable habitat following the allocation of the B50 full solution (S); *SD prop. - B50 S/EP* SD proportion of suitable habitat following the allocation of the B50 S for public lands (Extra-Property; EP). *SD prop. - B50 S/IP* proportion of suitable habitat following the allocation of the B50 S for private properties (Intra-Property; IP). *(%) Δ DS - B50 S* Increase in SD obtained by the allocation of the full B50 S; *(%) Δ DS - B50 S/EP* Increase in SD obtained by the allocation of the B50 S for public lands (Extra-Property; EP). *(%) Δ DS - B50 S/IP*; Increase in SD obtained by the allocation of the B50 S for private properties (Intra-Property; IP);

**Supplementary Table IV-9. Increment in species distribution following the restoration target allocation (scenario EW). See table footnote for column descriptions.**

| Species | SD prop. HQH - B100 | SD prop. HQH - EW S | SD prop. HQH - EW S/EP | SD prop. HQH - EW S/IP | (%) Δ in HQH - EW S | (%) Δ in HQH - EW S/EP | (%) Δ in HQH - EW S/IP | Species | SD prop. HQH - 100 | SD prop. HQH - EW S | SD prop. HQH - EW S/EP | SD prop. HQH - EW S/IP | (%) Δ in HQH - EW S | (%) Δ in HQH - EW S/EP | (%) Δ in HQH - EW S/IP |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Sp1** | 0.89 | 0.94 | 0.91 | 0.92 | 5.61 | 1.76 | 3.68 | **Sp32** | 0.91 | 0.93 | 0.92 | 0.92 | 3.07 | 1.38 | 1.54 |
| **Sp2** | 0.90 | 0.93 | 0.91 | 0.91 | 4.02 | 1.74 | 1.90 | **Sp33** | 0.43 | 0.47 | 0.44 | 0.46 | 9.09 | 1.91 | 7.58 |
| **Sp3** | 0.90 | 0.93 | 0.91 | 0.92 | 2.77 | 1.08 | 1.55 | **Sp34** | 0.67 | 0.73 | 0.69 | 0.70 | 7.85 | 3.21 | 4.06 |
| **Sp4** | 0.99 | 0.99 | 0.99 | 0.99 | 0.28 | 0.17 | 0.02 | **Sp35** | 0.71 | 0.76 | 0.73 | 0.73 | 7.06 | 2.91 | 3.72 |
| **Sp5** | 0.95 | 0.99 | 0.97 | 0.97 | 4.68 | 2.39 | 1.94 | **Sp36** | 0.67 | 0.73 | 0.69 | 0.70 | 7.85 | 3.21 | 4.06 |
| **Sp6** | 0.87 | 0.93 | 0.89 | 0.90 | 6.73 | 2.88 | 3.54 | **Sp37** | 0.81 | 0.88 | 0.84 | 0.85 | 9.28 | 3.99 | 4.87 |
| **Sp7** | 0.95 | 0.99 | 0.97 | 0.97 | 4.68 | 2.39 | 2.09 | **Sp38** | 0.61 | 0.68 | 0.62 | 0.67 | 11.87 | 2.19 | 9.46 |
| **Sp8** | 0.88 | 0.97 | 0.92 | 0.93 | 10.44 | 5.08 | 5.41 | **Sp39** | 0.82 | 0.89 | 0.85 | 0.85 | 8.14 | 3.55 | 4.07 |
| **Sp9** | 0.94 | 0.94 | 0.94 | 0.94 | 0.59 | 0.33 | 0.17 | **Sp40** | 0.87 | 0.89 | 0.88 | 0.88 | 2.88 | 1.40 | 1.40 |
| **Sp10** | 0.66 | 0.71 | 0.68 | 0.68 | 7.63 | 3.15 | 4.23 | **Sp41** | 0.97 | 0.99 | 0.99 | 0.98 | 2.29 | 1.54 | 0.37 |
| **Sp11** | 0.89 | 0.98 | 0.93 | 0.93 | 10.34 | 5.01 | 5.33 | **Sp42** | 0.81 | 0.87 | 0.83 | 0.85 | 8.28 | 3.48 | 5.07 |
| **Sp12** | 0.89 | 0.95 | 0.92 | 0.92 | 6.52 | 2.55 | 3.34 | **Sp43** | 0.97 | 1.00 | 1.00 | 0.97 | 3.43 | 3.16 | 0.02 |
| **Sp13** | 0.95 | 0.97 | 0.97 | 0.95 | 2.64 | 2.11 | 0.45 | **Sp44** | 0.70 | 0.75 | 0.72 | 0.73 | 7.97 | 3.41 | 4.23 |
| **Sp14** | 0.73 | 0.76 | 0.73 | 0.74 | 4.21 | 1.21 | 2.39 | **Sp45** | 0.72 | 0.78 | 0.75 | 0.75 | 8.08 | 3.40 | 4.39 |
| **Sp15** | 0.86 | 0.89 | 0.87 | 0.88 | 3.22 | 1.25 | 1.41 | **Sp46** | 0.82 | 0.83 | 0.82 | 0.82 | 2.04 | 0.77 | 0.60 |
| **Sp16** | 0.76 | 0.82 | 0.78 | 0.80 | 7.66 | 2.53 | 5.15 | **Sp47** | 0.77 | 0.84 | 0.80 | 0.81 | 8.66 | 3.58 | 4.69 |
| **Sp17** | 0.79 | 0.83 | 0.80 | 0.81 | 5.30 | 2.18 | 2.90 | **Sp48** | 0.98 | 1.00 | 1.00 | 0.99 | 4.27 | 2.12 | 1.97 |
| **Sp18** | 0.84 | 0.92 | 0.88 | 0.88 | 9.57 | 4.40 | 5.08 | **Sp49** | 0.95 | 1.00 | 0.98 | 0.97 | 5.26 | 2.84 | 2.24 |
| **Sp19** | 0.87 | 0.95 | 0.91 | 0.91 | 9.94 | 4.69 | 5.02 | **Sp50** | 0.90 | 0.94 | 0.92 | 0.92 | 4.32 | 1.72 | 2.36 |
| **Sp20** | 0.96 | 1.00 | 0.98 | 0.98 | 4.34 | 2.33 | 1.74 | **Sp51** | 0.64 | 0.69 | 0.66 | 0.67 | 7.39 | 2.85 | 4.19 |
| **Sp21** | 0.80 | 0.87 | 0.82 | 0.85 | 9.03 | 2.99 | 5.84 | **Sp52** | 0.72 | 0.78 | 0.75 | 0.75 | 8.49 | 3.82 | 4.81 |
| **Sp22** | 0.91 | 0.95 | 0.93 | 0.93 | 3.95 | 1.81 | 1.97 | **Sp53** | 0.63 | 0.67 | 0.64 | 0.65 | 7.11 | 2.97 | 3.89 |
| **Sp23** | 0.92 | 0.99 | 0.96 | 0.96 | 8.16 | 3.96 | 3.96 | **Sp54** | 0.86 | 0.89 | 0.87 | 0.87 | 3.57 | 1.62 | 1.95 |
| **Sp24** | 0.83 | 0.91 | 0.87 | 0.87 | 9.36 | 4.15 | 4.84 | **Sp55** | 0.91 | 0.94 | 0.92 | 0.93 | 3.99 | 2.01 | 2.17 |
| **Sp25** | 0.90 | 0.95 | 0.92 | 0.93 | 5.54 | 2.34 | 2.97 | **Sp56** | 0.87 | 0.89 | 0.88 | 0.88 | 3.21 | 1.40 | 1.73 |
| **Sp26** | 0.90 | 0.95 | 0.92 | 0.93 | 5.23 | 2.34 | 2.66 | **Sp57** | 0.83 | 0.86 | 0.84 | 0.84 | 3.70 | 1.43 | 2.12 |
| **Sp27** | 0.88 | 0.92 | 0.89 | 0.90 | 4.76 | 2.17 | 2.66 | **Sp58** | 0.61 | 0.66 | 0.63 | 0.63 | 7.27 | 2.87 | 3.80 |
| **Sp28** | 0.67 | 0.73 | 0.69 | 0.70 | 7.85 | 3.21 | 4.06 | **Sp59** | 0.89 | 0.98 | 0.93 | 0.94 | 10.63 | 5.15 | 5.31 |
| **Sp29** | 0.68 | 0.73 | 0.70 | 0.70 | 6.53 | 2.72 | 3.35 | **Sp60** | 0.74 | 0.81 | 0.77 | 0.78 | 8.61 | 3.43 | 4.58 |
| **Sp30** | 0.63 | 0.67 | 0.64 | 0.65 | 7.11 | 2.97 | 3.89 | **Sp61** | 0.77 | 0.84 | 0.80 | 0.80 | 9.06 | 3.79 | 4.91 |
| **Sp31** | 0.86 | 0.94 | 0.90 | 0.91 | 10.03 | 4.59 | 5.59 | **Sp62** | 0.65 | 0.71 | 0.67 | 0.68 | 8.09 | 3.17 | 4.26 |

*Original SD prop.* = Species distribution (SD) proportion of remaining suitable habitat prior to the restoration target allocation, calculated using Zonation; *SD prop. - EW S* = SD proportion of suitable habitat following the allocation of the EW full solution (S); *SD prop. - EW S/EP* SD proportion of suitable habitat following the allocation of the EW S for public lands (Extra-Property; EP). *SD prop. - EW S/IP* proportion of suitable habitat following the allocation of the EW S for private properties (Intra-Property; IP). *(%) Δ DS - EW S* Increase in SD obtained by the allocation of the full EW S; *(%) Δ DS - EW S/EP* Increase in SD obtained by the allocation of the EW S for public lands (Extra-Property; EP). *(%) Δ DS - EW S/IP*; Increase in SD obtained by the allocation of the EW S for private properties (Intra-Property; IP);

# Chapter V
# Synthesis

# 1. Summary

This thesis aimed to advance the knowledge about the potential of the Brazilian Forest Code (BFC) enforcement for the conservation of old- and regrowing forests in the Brazilian Amazon. Specifically, I focused on the Legal Reserve (LR) requirement in private landholdings. LRs concentrate over one third of the forest cover in Brazil (Metzger et al. 2019), safeguarding the provision of ecosystem services and biodiversity conservation. Despite of the 2012 BFC relaxed conditions for law compliance, the Amazon biome still accumulates extensive LR deficits. The law allows off-site compensation of deficits, but, dependent upon the BFC regulation, its implementation could lead to local forest restoration where LRs fall short. Therefore, stickiness to the BFC is key for forest conservation and restoration in the Brazilian Amazon. Still, the 2012 BFC has been ineffective in promoting forest restoration (Azevedo et al. 2017), and additional policies such as the PLANAVEG or the PCI in Mato Grosso, will play an important role in supporting forest restoration through the BFC implementation. Previous studies mapped LR deficits across private properties in the Amazon, providing valuable information for the identification of candidate areas for forest restoration in private properties. Yet, the scope of previous estimates was limited to old-growth forests, and the potential contribution of ongoing forest regrowth for LRs demarcation and for the implementation of regularization mechanisms established by the BFC remained unknown. Also unknown was the location of private lands with priority for local restoration of LRs, and the amount of regrowing forests with high value for conservation. Overall, this thesis addressed these and other related shortcomings combining a variety of approaches, contributing to the discussion on regrowing forests governance in the Amazon biome. I addressed three research questions in three core research chapters (II, III and IV) and one Appendix Chapter. In the following paragraphs I summarize the main findings with respect to each research question.

**Research Question I -** What were the spatio-temporal patterns of net forest cover change in the Brazilian Amazon over the last decades for different tenure categories?

Chapter II uncovered distinct forest cover change (FCC) patterns, and the associated carbon balance, across tenure categories, between 1985 and 2012, for the influence area of the *Cuiabá-Santarém* (BR-163) highway (*objective 1*). Complementarily, the Appendix Chapter tracked annual emissions from deforestation and forest fragmentation for the same timeframe as Chapter II. The Appendix Chapter also contributed methodologically to

Chapter II, detailing and providing the carbon bookkeeping model used to analyze the FCC carbon balance.

The study area is highly dynamic, and the analysis provided rich insights about how FCC may have responded to socio-economic and policy changes that occurred in the last decades. In the 27 years analyzed, the influence area of the *Cuiabá-Santarém* highway accumulated nearly 40% of old-growth forests clear-cut deforestation. Forest losses were more intense in private properties identified using the CAR dataset, largely exceeding the threshold established by the BFC, both in Mato Grosso and Pará. Deforestation rates remained high in undesignated public lands of Mato Grosso and Pará. This pattern was interpreted as an indicative that anti-deforestation policies (i.e., PPCDAm) were more effective in private properties than in public lands. In Mato Grosso, most old-growth forests remaining in 2012 were located in private properties and lands with unknown tenure, while in Pará, the bulk of remaining old-growth forests was located in undesignated and designated public lands.

Old-growth forests deforestation dominated the carbon balance across tenure categories. Prevailing net deforestation showed that a relevant turnaround from forest loss to forest gain did not take place in any of the land tenure categories across time. Carbon sequestration was also low, due to the short life cycle of regrowing forests and decreasing prevalence of forest regrowth on previously deforested lands, indicating a land use intensification process. In addition, the Appendix Chapter supplemented *objective (1)* and showed how landscape deforestation patterns influenced carbon emissions in edge-affected areas. Edge emissions were low compared to old-growth deforestation carbon losses, but their proportional relevance increased during the study period as a consequence of the overall decline in deforestation rates during the 2000s.

**Research Question II -** What are the outcomes of different Brazilian Forest Code implementation assumptions for the protection of current stocks of old- and regrowing forests?

Chapter III quantified the contribution of old- and regrowing forests for the demarcation of LRs in private landholdings overlapping the Amazon biome (in 2014). This chapter also estimated the protection status of old- and regrowing forests in private lands and their eligibility for issuing forest certificates for off-site compensation of LR deficits.

Results approximated the potential of regrowing forests for reducing LRs deficits on-site. The analysis conducted in Chapter III showed that, contingent on the interpretation of the

BFC, a large extent of regrowing forests (6.3 Mha), located in properties with LR deficit, could be protected. Including current stocks of regrowing forests in LRs demarcation could reduce over one third of forest deficits across analyzed properties. This potential contribution was larger for states at the arc of deforestation, i.e., Mato Grosso, Pará, Rondônia, Tocantins, and Maranhão. In addition, around one fourth of areas deforested after 2008 are under regrowth, and if the illegality of these clearings is confirmed, such regrowing forests must be protected.

Chapter III explored the contribution of forest certificates issued from old- and regrowing forests, to be traded in a market mechanism, for LRs deficits off-site compensation. The inclusion of regrowing forests stocks increased certificates availability by almost one third. Alternative scenarios for market regulation were explored to address the possible outcomes of forest certificates trading, for old- and regrowing forests protection additionality (and associated carbon stocks), and for law compliance levels. Depending on the regulatory setup, LR deficits could be entirely compensated by the certificate trading mechanism, without the need for additional restoration. Compromises between old-growth forests protection additionality and law compliance were large, depending on the eligibility of certificates issued from protected forests and the geographical range of the market. As a consequence, permissive regulations could leave large stocks of old-growth forest carbon unprotected. The carbon quantification indicated that carbon stocks (in 2014) in protected and unprotected regrowing forests in private landholdings are negligible in comparison to stocks of old-growth forests. Therefore, BFC regulations that facilitate the assimilation by the compensation mechanism of certificates issued from carbon rich, and unprotected old-growth forests, are desirable. However, under the assumptions made for each regulatory setup, the inclusion of regrowing forests certificates did not affect the demand for unprotected old-growth forest certificates.

**Research Question III –** How do the costs and benefits of allocating forest restoration to private lands vary under different prioritization scenarios, and how do they compare with the outcomes of restoration in public lands?

Chapter IV quantified the potential costs and benefits of recovering a share of Mato Grosso's LR forest restoration target. A prioritization algorithm was applied to rank suitable candidate areas for forest recovery, combining four criteria: habitat enhancement for selected mammal species, carbon storage maximization, likelihood of spontaneous forest regeneration, and minimization of agricultural opportunity costs (*Objective IV*). Different scenarios were analyzed, ranging from a full prioritization of habitat enhancement to an "equal weights" scenario, in which all criteria received the same importance. The model allocated forest restoration at the pixel level, allowing a posterior comparison of costs and benefits. Restoration (1Mha) was allocated to private lands with deficits (as identified by Chapter III). Costs and benefits were compared with those of the allocation of the same amount of restoration (1Mha) to public lands (*Objective V*). The quantification of carbon storage enhancement and the total costs associated with forest restoration, accounted for the incremental allocation of restoration targets across 11 years (with an end year in 2030 – the same as the expected conclusion of the state restoration strategy).

The results showed substantial variation in the spatial distribution of priority areas for restoration across Mato Grosso and between scenarios. The scenario fully prioritizing biodiversity distributed high-rank areas more evenly across space, except for high-value clusters, mostly overlapping areas of species geographical range that are small and heavily deforested. Leveling criteria' weights concentrated priority areas in northwestern Mato Grosso, where there is an abundance of old-growth forests patches, carbon densities are higher and opportunity costs are smaller. Mean habitat gains were very similar across scenarios, but trade-offs in habitat enhancement occurred between species for different scenarios. Species demanding large habitats benefited less from restoration allocation in all the scenarios. Carbon sequestration cost-effectiveness (in private lands with LR deficits) nearly doubled when carbon storage and opportunity costs minimization were assigned the same weights as habitat enhancement (equal weights scenario).

Allocation of forest restoration to landholdings with LR deficits was more beneficial for species with highly deforested habitats, due to the concentration of private lands in areas where deforestation is widespread. On the other hand, the results showed a potential for carbon sequestration at lower costs in public lands.

The inclusion of regrowing forest areas increased the cost-effectiveness of carbon sequestration in scenarios balancing criteria' weights and in public lands. The share of forest regrowth ranged between 40% to 80% of the areas selected as priority.

## 2. Main conclusions and implications

The main conclusion and implications regarding **Research Question I** (Chapter II):

*The influence area of the Cuiabá-Santarém highway is not near experiencing a turnaround from net forest losses to net forest gains.*

Cutting back emissions from LULCC is a major pillar of Brazil's NDC, which targets zero net deforestation in the Amazon and the restoration of 12Mha of forests countrywide by 2030. However, the net forest cover change patterns quantified indicate that the demand for land, either for speculative or productive uses, remains high in the *Cuiabá-Santarém* focus region. These circumstances conflict with Brazil's commitments to reduce emissions from the LULCC sector. Such commitments are not simple to achieve and in this specific case, business as usual land use dynamics will not result in forest expansion.

Even though land use dynamics are context specific and the results apply only to the *Cuiabá-Santarém* study case, this work demonstrated that long-term information on FCC, with a high spatio-temporal resolution, is important to identify areas where old- and regrowing forests are threatened the most. Moreover, in combination with knowledge about the spatial distribution of tenure regimes, information about FCC dynamics provides insights about the processes influencing the decision to (re)clear forests or allow forest regrowth, shedding light on actor-specific responses to past and current land use policies. The results hinted to where recent anti-deforestation policies were more effective, and where improvement is necessary, and reinforce that, the need for better forest governance to promote forest restoration applies to all tenure regimes. Specifically, future conservation and restoration policies in Mato Grosso should particularly target the BFC enforcement in private properties. If the hypothesis that the lands with unknown tenure are mostly private landholdings to be registered with the CAR is confirmed, then the areas where the BFC regulations should apply accounted for 83.6% of the accumulated clear-cut deforestation in the study area. This shows that historically, the BFC alone could not guarantee forest protection in private lands, and the continuity of public (e.g., credit restriction, raids and levying fines) and private (e.g., beef and soy moratoria) anti-deforestation policies is necessary to support the law enforcement. On the other hand, in southwestern Pará, to prevent future deforestation is urgent to clarify tenure, assign public undesignated lands to conservation or protection purposes, and improve current conservation units' management. This information is in line with findings by other studies (Azevedo-Ramos and Moutinho 2018; Freitas et al. 2018; Sparovek et al. 2019) which argue that the extensive undesignated public lands in the Brazilian Amazon are putting massive forest and carbon stocks at risk of deforestation. Beyond carbon, the lack of use designation in public lands threatens the viability of large shares of endangered species populations (Freitas 2019).

The main conclusions and implications regarding **Research Question II (Chapter III)**

*It is necessary to substantially improve regrowing forests governance.*

If the rules established for the protection of native vegetation are valid for forests undergoing regeneration, then the amount of forest regrowth qualifying as protected in private lands would exceed PLANAVEG's restoration target for the Amazon biome. However, the conditions for effectively protecting second-growth forests are barely existent, and several measures are necessary to improve their governance. First, legal frameworks should be complemented to clarify the criteria for identifying regrowing forests to be protected (Vieira et al. 2014). Currently, the BFC provides a loose definition of native vegetation, which includes forests undergoing succession, but offers no additional standards. The revision of the states' environmental regularization programs (the PRAs), conducted in Chapter III, also revealed that regrowing forests governance was not improved by states' legislations. Therefore, it should be made clear if forests regrowing in properties with less LR than required by the law are automatically protected by the BFC, and if they are not, how to identify those which are worthy of protection due to their potential for ecosystems recovery, proposing scientifically sound and operational indicators (Aronson et al. 2011). It is also important that regulations take biophysical and agricultural systems variation into account, to avoid imposing unfair requirements on producers that rely on forest fallows or have small areas available for subsistence agriculture (Cronkleton et al. 2013; Román-Dañobeytia et al. 2014). Second, a remote sensing-based monitoring program, consistently providing indicators for forest regrowth dynamics, is lacking. Clear-cut deforestation in the Amazon is monitored annually by the PRODES system, but TerraClass, the official program mapping forest regrowth (among other land cover classes), has not published maps since 2014, and it is not clear if the project will continue. Ideally, regrowing forests monitoring should be methodologically consistent with PRODES, as this product is officially used to support policy elaboration and assessment. Nevertheless, in the absence of official estimates, the recently launched MapBiomas land monitoring program (http://mapbiomas.org) could be an alternative source of annual FCC maps. Third, such FCC maps, in combination with the CAR, can enable tracing high-value second growth forests (e.g., of advanced age, or key for landscape connectivity) to specific landholders to support the application of negative and positive enforcement measures. For example, the soy and beef moratoria ban producers associated with recent deforestation. This ban could be extended to exclude farmers not complying with the BFC, e.g., those who (without license) clear old-growth and high-value second growth forests that could be used to demarcate LRs, or those who are not engaging in restoration or off-site compensation. So far, the CAR has been underused for law enforcement and other policies, largely due to the slowness by successive governments in finalizing the public registry (Azevedo et al. 2017). In fact, researchers argue that the cadastre has been used to proof the existence of forest surplus in properties, to support issuing licenses for legal deforestation (Carvalho et al. 2019b). Therefore, the CAR should be completed and the state's PRAs implemented, so that the perception of impunity is no longer prevailing among farmers and restoration targets can be addressed through law compliance.

*In a worst-case scenario, the enforcement of the BFC would not drive legal reserve forest restoration (apart from deficits associated with post 2008 deforestation) and additional old-growth forests protection.*

The amount of protected old-growth forests apt for issuing forest certificates, surpasses the extent of LR deficits that qualify for off-site compensation. It is plausible to speculate that, dependent upon the market regulations defined by the states' PRAs, the large offer of protected certificates might inhibit both on-site restoration and the purchase of certificates issued from unprotected old-growth forests. However, it is necessary to make a few remarks. First, as discussed in Chapter III, issuing and negotiating forest certificates requires secured ownership, which is highly uncertain in the Amazon (Brito 2017), and could drastically reduce certificates' offer. The CAR itself attests that conflicting land ownership is widespread, as it includes multiple overlapping landholdings (and landholdings overlapping other tenure categories). Second, the analysis conducted in Chapter III does not consider the administrative costs of engaging in the market, which could make issuing certificates unappealing if their prices are too low (Soares-Filho et al. 2016). Ignorance to the BFC mechanisms may also reduce certificates' offer. Finally, the pathways that landowners will choose to comply with the BFC are uncertain. Recent empirical research suggests that, if the law is enforced, on-site LR recovery might be the preferred option by most farmers (Pacheco et al. 2017), especially if the local support to conduct restoration is available (Daugeard 2018). Hence, future investigations on the potential behaviour of farmers towards strategies for the BFC compliance are desirable. In addition, a proposal for a national policy regulating payments for ecosystems services in Brazil (PNPSA, acronym in Portuguese) has recently passed the congress and will be analysed by the senate (PL 312/2015, approved by the congress in September 2019). The PNPSA proposal includes the offer of unprotected forest certificates in a national cadastre of environmental services, opening the possibility for interested parties, beyond farmers with LR deficits, to pay for forests surplus protection. This could include (national and international) individuals or companies seeking to compensate environmental liabilities, non-profit organizations, or even governments at various levels. Hopefully, this mechanism would bring additional protection to forest surpluses, if the trade of certificates for BFC compliance alone fails to do so. Nonetheless, the estimates provided by this work showed that conservative regulations could help to maximize multiple benefits from law enforcement.

The main conclusions and implications regarding **Research Question III (Chapter IV)**

*Cost-effectiveness of forest restoration can be significantly increased depending on the scenario, but large-scale forest expansion is still costly.*

Chapter IV identified areas of strong synergy between the desired outcomes of forest restoration, an important starting point in large-scale forest restoration. Still there are many challenges to be addressed, related to costs, capacitation, and land governance. This work focused on the identification of areas apt for natural forest regeneration, since it usually incurs in less monetary costs in comparison to active restoration (Holl and Aide 2011). Even though cost-effectiveness of restoration increased as feasibility criteria received higher weights, the opportunity and direct costs involved are still high in Mato Grosso, likely requiring auxiliary policies to enable LRs recovery. Here, the proposed PNPSA could become an important policy, as it foresees economic incentives, beyond what is currently existing (Moreira et al. 2016), for farmers recovering LRs located in priority areas. This includes a facilitated access to credit lines with low interest rates, tax exemption, technical assistance in consortium with municipalities, among other important initiatives (PL 312/2015). Such benefits could help reduce direct costs of restoration considerably. However, the proposed PNPSA vetoes monetary payments for LRs recovery, which could make the program less appealing to farmers.

There is a culturally rooted perception among farmers that early stage regrowing forests are a waste of productive land, and their shrubby aspect may suggest land abandonment and failure in restoration (Chazdon and Guariguata 2016; Zahawi et al. 2014). In this context, technical assistance by ecological restoration experts is fundamental to help farmers overseeing healthy forest recovery trajectories (Chazdon and Guariguata 2016), increasing the chances of second-growth forests persistence. Also, the long term economic and environmental benefits provided by second-growth forests are difficult to monetize, and farmers may be compelled to engage in activities providing fast economic return (Chazdon and Guariguata 2016) – especially in the context of weak law enforcement. However, if restoration costs are too high, other strategies to enable forest restoration can also be pursued. For example, mixing native and non-native commercial species in restoration can provide win-win situations, supporting species diversity recovery and compensating costs through harvesting of sustainable timber at the same time (Brancalion et al. 2019). Finally, in Chapter IV, the CAR layer was used to identify candidate areas with LR deficit for private lands forest restoration. However, the CAR layer does not inform about ownership security, meaning that the number of candidate properties for LR restoration under the PCI program – and eligible for receiving benefits in the context of a PES program - could have been overestimated by this work. Therefore, to provide better estimates and support restoration planning in Mato Grosso (and other regions in Brazil) it is important to clarify land tenure and include information about land titling availability on the property registry.

*Land tenure should be accounted for in detail by restoration prioritization assessments*

Decision support systems aiding forest restoration planning can benefit from an accurate representation of land tenure, but this is hardly the standard for restoration assessments (McLain et al. 2018). The PCI strategy has a specific focus in forest restoration on private lands with BFC deficits, therefore the spatial representation of tenure was necessary. This work showed that the attainable benefits and costs of restoration differed strongly across private and public lands, a solid argument for the inclusion of detailed information on tenure and governance instruments in such assessments. Results also demonstrated the enormous potential of the CAR to support conservation and restoration planning in the Amazon. Future assessments would benefit very much from the continuous improvement of the CAR dataset, especially concerning data validation. Similarly, the potential of forest restoration in public lands is poorly addressed by research. In Mato Grosso this potential was non-negligible, even though private ownership is the dominant tenure regime. It is important that future research further investigates this potential for states where public lands prevail, like Pará, and support the formulation of policies to advance restoration in state owned lands (Assunção and Gandour 2019).

# 3. Final remarks and outlook

The results of this thesis advanced the knowledge about the potential of the current legal framework for the protection of old- and regrowing forests in the Brazilian Amazon, and for achieving targets set by policies related to the BFC. This work showed that there is room for substantial carbon and biodiversity enhancement through the enforcement of the BFC. It also brought to light the importance of discussing regrowing forests governance, and the vulnerabilities of current legislations. However, despite its merits, there are important shortcomings to be discussed. In the following paragraphs I will address some key limitations of the research presented by the core chapters, highlighting potential alternatives for its improvement by future studies. I then discuss research transferability potential and close this synthesis chapter with a policy outlook.

In all the three core chapters (II-IV) I used carbon storage and sequestration as one example of an ecosystem benefit provided by forest conservation and restoration. However, in the future, the carbon modelling could be improved, especially the carbon sequestration assessment. I estimated annual carbon uptake by forest regrowth based on a fixed annual recovery rate of 1.2% of the original carbon stocks (Lennox et al. 2018). However, research has found that past land use management and biophysical characteristics (i.e., temperature, rainfall, soil) drive the potential for forest carbon uptake, which is spatially heterogeneous. In this case, a spatially explicit map of carbon sequestration potential by regrowing forests, similar to Chazdon et al. (2016b), would improve the annual carbon balance estimates and could also be used as an additional criterion in the restoration prioritization framework presented here.

Recent studies have started to investigate current changes in forest regrowth processes, associating variations in secondary vegetation dynamics observed between 2004 and 2012 to an influence of the PPCDAm (Carvalho et al. 2019a; Gandour 2018). However, these studies relied on TerraClass, which covers only the recent period of LULCC dynamics in the Amazon, and offers maps at spaced time intervals (1991, 2000, 2004, 2008, 2010, 2012, and 2014). The lack of a longer time series hampers the establishment of a baseline from which to analyse forest regrowth dynamics and draw associations with land use policies. In this context, the FCC analysis conducted by chapter II showed a highly dynamic behaviour of forest regrowth for different tenure categories, and suggests the need for extended time series, with high spatio-temporal resolution, to support policy assessments. However, the conclusions drawn are restricted to the influence area of the Cuiabá-Santarém highway. In the future, extended time series of FCC, similar to the one employed by this study, but covering the whole Amazon, would allow to assess the impacts of the PPCDAm (and other policies) on forest regrowth dynamics. The LULCC maps published by the Mapbiomas project (https://mapbiomas.org) cover the year of 1985 onwards annually, with a 30 m spatial resolution, and could be used for such analysis.

Chapter III provided the first quantification of regrowing forests contribution to LRs demarcation, among other important estimates that can enrich the debate about regrowing forests governance in the Amazon. However, there are gaps to the CAR data that add considerable uncertainties to the LR deficits numbers provided. This is because the CAR data does not allow to link one owner to multiple properties. Therefore, each registry entry was treated as a unique ownership, when, in fact, multiple ownerships by one individual are not uncommon. Therefore, it was not possible to identify cases in which a farm with forest surplus offsets the LR deficits of another farm of the same owner. In this case, an ownership disclosure, perhaps using an identification number in the dataset, would be important to refine the estimates.

Another important shortcoming was the absence of estimates for PPA deficits, since the spatial resolution applied was insufficient to capture fine scale buffers that should be set for protection or recovery of riparian areas. Buffer sizes depend on the water stream width, an information that is also lacking, challenging the calculation of PPAs deficits. Still, in Chapter IV, the likelihood map used as criterion includes proximity to drainage as a driver of forest regeneration. Hence, the LR recovery allocation simulated by this chapter favoured river buffers, likely allocating forest expansion to deforested PPAs – which is in accordance with the 2012 BFC, which allows the deduction of LR deficits through the recovery of PPAs (Brancalion et al. 2016a). However, I did not quantify the extent of LRs recovery simulated on water streams buffer zones, which could be done in future assessments.

Chapter IV made an important contribution to forest restoration planning state of the art, by proposing a decision support tool that includes detailed tenure information, and restoration demands expected from law enforcement. However, future versions of this model could be improved. First, proximity to old-growth forests is associated with high priority values for forest restoration. However, even though the analysis conducted in Chapter IV assumes that the BFC will be enforced (i.e., LRs are protected) it does not account for the risk of future illegal and legal deforestation in Mato Grosso, taking place on public lands and private properties with forest surplus. Future old-growth forests clear-cut occurring near restoration sites could negatively influence forest succession, e.g., by impacting the fauna and reducing seed availability, and should be accounted for by restoration planning. One way around this limitation could be to incorporate future scenarios of LULCC to the prioritization assessment. Still regarding Chapter IV, the representation of biodiversity can also be improved. Even though the prioritization model innovates for including the landscape requirements (i.e., home range) for multiple species, habitat constraints (e.g., altitude, forest type), relevant for specifying the species suitable range, were not included. In this regard, outputs from species distribution models could be an asset to this model and are compatible entries for Zonation (Budiharta et al. 2018; Thomson et al. 2009).

Given that the CAR is available for rural areas countrywide, the prospects for research transferability are good. In Chapter IV, Mato Grosso was chosen as a study case for the prioritization model because it has a forest restoration target and policy, aligned with the BFC enforcement. However, a prioritization assessment for other states in the Amazon, could provide important information to support restoration planning, especially for Pará, Maranhão, Tocantins, and Rondônia. The prioritization model could also be easily adapted to other biomes. In the Atlantic forest, several other models have been proposed (Molin et al. 2018; Strassburg et al. 2019), but an analysis similar to what was conducted here could help to investigate the impact of using property boundaries as constraints in forest restoration simulation. Finally, it is also crucial to address restoration of non-forest ecosystems. Researchers and practitioners argue that the efforts to promote landscape restoration under the Bonn Challenge have been neglecting non-forest ecosystems, which are also crucial for climate change mitigation, water quality conservation, and to reverse biodiversity losses (Temperton et al. 2019). Despite of their high levels of biodiversity and crucial role in providing ecosystem services, non-forest ecosystems in Brazil have been suffering with the conversion of native vegetation at alarming rates, driven by agriculture, infrastructure, and urban expansions (Lapola et al. 2013), in combination with a small protected areas coverage, low protection levels by the BFC, and weak law enforcement (Overbeck et al. 2015; Strassburg et al. 2017). It is urgent to expand the successful policies implemented in the Brazilian Amazon to other ecosystems, especially to the *Cerrado*, a highly threatened biodiversity hotspot (Strassburg et al. 2017). Therefore, future studies aiming to identify prime areas for conservation and opportunities for the restoration of natural grasslands, wetlands, and savannah ecosystems in Brazil are highly needed (Strassburg et al. 2017).

In the mid-2000s concerted policies improved environmental governance in the Brazilian Amazon, successfully reducing deforestation rates by 84% between 2004 and 2012. Negative enforcement actions typical of a frontier governance (Nepstad et al. 2014), such as field raids, blacklisting, and the moratoria, were crucial to control deforestation. However, policies focusing on tenure regulation and positive enforcement, could potentially pave the way for a change in the underlying rationale of farmers, towards a model that integrates conservation and restoration to agricultural production. Some examples of important policies were mentioned throughout this thesis, such as the Terra Legal program, the national REDD+ strategy, the PLANAVEG, and, hopefully, the future PNPSA. The Brazilian Forest Code crosscuts all these policies, that combined, could drive the Amazon through a "State Forest Policy" transition pathway. Nevertheless, the general trend of increase in deforestation rates since 2012 in the Amazon frustrates this expectation. Persistency and the recent upsurge in deforestation can be associated with gaps in many of the policies in place (Carvalho et al. 2019b; Gollnow et al. 2018; Klingler et al. 2017). For example, every year Brazil misses the opportunity of collecting millions of dollars from fines linked to environmental infractions, which could be applied to conservation and restoration programs in the Amazon, including the implementation of the BFC (Carvalho et al. 2019b). The delay in implementing the 2012 BFC could also have put the law in discredit, and farmers likely do not fear sanctions from not engaging in LRs and PPAs restoration. However, the economic, political, and moral crises installed since 2013 have dramatically changed the course of the Brazilian environmental agenda. Attacks to environmental legislation and policies were intensified in 2017, when a series of decrees were signed by the former President Michel Temer, downsizing protected areas, reducing environmental licensing requirements, and facilitating squatters' land claims over illegally deforested areas (Crouzeilles et al. 2017a). In 2018, the election of President Jair Bolsonaro aggravated this situation. Since January 2019, the new President has put forward several attacks to human rights and environmental policies, deepened budget cuts for environmental policies and science, shamed conservationists and NGOs, and threatened to withdraw Brazil from the Paris Agreement (Abessa et al. 2019; Artaxo 2019). The prospects for the implementation of the PLANAVEG are also not good. Bolsonaro's administration abolished the CONAVEG (Decree 9759, 11th of April 2019), the national council that had been created to put the native vegetation restoration plan into practice, and which also made a bridge with the civil society. The austere political climate has also facilitated the submission of several projects of laws, including one that directly attacks the BFC, suggesting the extinction of the Legal Reserves requirement in private properties

(Abessa et al. 2019; Metzger et al. 2019). The current President's anti-conservation discourses and stands have been directly linked to the registered increase in fires and deforestation in 2019, when registered clear-cut rates were the highest since 2008. Consequently, the Amazon Fund suffered sanctions from Germany and Norway, its largest donners. The Amazon Fund is the major financer of the Brazilian REDD+ policy, and destined resources to support the BFC implementation.

For all the services they provide, Legal Reserves should be considered assets and not burdens, and their protection and restoration are of common interest (Metzger et al. 2019). Failing to acknowledge their importance would represent a missed opportunity for climate change mitigation, biodiversity conservation, and the creation of new pathways for socioeconomic development. Hopefully, in the near future, Brazil will take back its previous role of environmental leadership, and base the country's policies on sound science, consistent with the conservation and restoration commitments it has ratified, and with the goal of alleviating social and environmental historical debts (Loyola 2014).

# Appendix

# Historical carbon fluxes in the expanding deforestation frontier of Southern Brazilian Amazonia (1985-2012)

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# Abstract

In tropical areas, pioneer occupation fronts steer the rapid expansion of deforestation, contributing to carbon emissions. Up-to-date carbon emission estimates covering the long-term development of such frontiers depend on the availability of high spatial-temporal resolution data. In this paper we provide a detailed assessment of carbon losses from deforestation and potential forest degradation from fragmentation for one expanding frontier in the Brazilian Amazon. We focused on one of Amazonia's hotspots of forest loss, the BR-163 highway that connects the high productivity agricultural landscapes in Mato Grosso with the exporting harbors of the Amazon. We used multi-decadal (1984-2012) Landsat-based time series on forested and non-forested area in combination with a carbon book-keeping model. We show a 36% reduction of 1984s biomass carbon stocks, which led to the emission of 611.5 $TgCO_2e$ between 1985-1998 (43.6 $TgCO_2e\ yr^{-1}$) and 959.8 $TgCO_2e$ over 1999-2012 (68.5 $TgCO_2e\ yr^{-1}$). Overall, fragmentation related carbon losses represented 1.88% of total emissions by 2012, with an increasing relevance since 2004. We compared the Brazilian Space Agency deforestation assessment (PRODES) with our data and found that small deforestation polygons not captured by PRODES had increasing importance on estimated deforestation carbon losses since 2000. The comparative analysis improved the understanding of data source related uncertainties on carbon estimates and indicated disagreement areas between datasets that could be subject of future research. Furthermore, spatially explicit, annual deforestation and emission estimates like the ones derived from this study are important for setting regional baselines for REDD+ or similar payment for ecosystem services frameworks.

## 1. Introduction

Conserving Amazonia's massive forest carbon stocks is a central objective of the climate change mitigation debate. Even more, recent assessments point that it is important to recover large tracts of forest in the Amazon to promote carbon sequestration and restore the hydric regional balance (Nobre 2014). However, since the early 1960s the Brazilian Amazon has lost nearly 18% of pristine forest, over 760,000 km$^2$ to clear cutting deforestation alone (INPE 2014b). As a consequence, deforestation in the Brazilian Amazon was a net source of 0.10 to 0.15 Pg C yr$^{-1}$ during the 2000s (Aguiar et al. 2012), accounting for 12% of the global land use and land cover change (LULCC) emissions, and 1.5% of overall global $CO_2$ emissions in 2009 (Le Quéré et al. 2009; Malhi 2010; Pan et al. 2011). Worthy to mention, Brazil's last decade's efforts to halt deforestation in the Amazon already pushed down emission rates, even though the agricultural frontier is moving towards higher biomass density areas (Ometto et al. 2014). Still, we are far from affirming that deforestation will soon come to an end (Fearnside 2015), and even further away from a scenario of large-scale forest restoration (Aguiar et al. 2016).

Despite its importance LULCC is the most uncertain component of the global carbon budget (Houghton et al. 2012). Different approaches have been proposed to deliver estimates, i.e., book-keeping models, dynamic vegetation models and earth system models (Pongratz et al. 2014). Carbon book-keeping models have been used in the past to quantify the effects of land use change on carbon stocks (Achard et al. 2004; DeFries et al. 2002; Houghton et al. 2000). They rely on the availability of datasets describing land use and land cover trajectories to allocate carbon losses and gains temporally (Ramankutty et al. 2007), of information on biomass density and of knowledge to characterize carbon decay under different land management systems (Aguiar et al. 2012). Global or biome level studies depend on large datasets, often with moderate spatial resolution (Song et al. 2015) or temporal extent (Hansen et al. 2013) and aggregated census data (Imbach et al. 2015; Leite et al. 2012). Regional scale assessments, on the other hand, can overcome methodological constraints, bringing insights to specific processes relevant to the carbon budget (Carlson et al. 2012; Toomey et al. 2013). On this matter, efforts to produce spatial-temporal information and modeling tools able to accurately represent and quantify emissions are critical (Asner et al. 2010). Newly available Landsat derived datasets covering deforestation since the early 1980s (Müller et al. 2016a) provide the detail necessary to characterize long term land use change and spatially heterogeneous carbon decay processes, such as emissions from biomass mortality due to edge effects and forest fragmentation (Numata et al. 2010). This is fundamental for understanding the long-term development of deforestation frontiers and the contribution of forest degradation to emerging emission patterns.

Pervasive land uses cause different levels of forest loss, with clear cut deforestation being the most extreme. Forest fires and selective logging degradation also create gradients of forest loss and often develop positive feedbacks that intensify disturbances and carbon stocks depletion (Aragão et al. 2014). Although forest degradation affects only a percentage of forest carbon stocks this process spanned an area almost two times the clear-cut extension across the Brazilian Amazon between 2007-2013 (INPE 2014a) and could represent up to 47% of deforestation gross emissions (Aguiar et al. 2016). Another driver of forest degradation often neglected in carbon assessments is forest fragmentation leading to edge effects (Broadbent et al. 2008) which also interacts with other degradation processes (e.g., understory fires). Increased exposure to adverse conditions (e.g., wind turbulence, light exposure, increased dryness) cause tree mortality and canopy gaps formation, potentially altering forest structure at edges, leading to aboveground biomass collapse (Berenguer et al. 2014; de Paula et al. 2011; Laurance et al. 1997; Nascimento and Laurance 2004).

In this study, we attempt to understand and quantify the impact of emerging spatial-temporal deforestation and edge emission patterns over almost 30 years along a deforestation frontier area, the influence area of the BR-163 highway, in southeastern Amazonia. Focusing on this area, we estimate historical carbon emissions from clear-cut deforestation and edge biomass collapse using the book-keeping modeling framework INPE-EM (Aguiar et al. 2012), here augmented to assess emissions from forest fragmentation (Numata et al. 2010; Numata et al. 2011). We used annual deforestation maps between 1984-2012 with 30 m spatial resolution (Müller et al. 2016a) as inputs, to derive spatial and temporal dynamics of forest edges. Specifically, the objectives of this study were to:

(1) Quantify potential carbon emissions from deforestation and edge creation for the period of 1985-2012;

(2) Uncover trends and spatial-temporal variability of biomass losses and carbon emissions;

(3) Quantify the impact of using different deforestation assessments, Mueller et al. (2016) and the Brazilian Institute for Space Research deforestation assessment (PRODES) (INPE 2014b) on estimated carbon emissions.

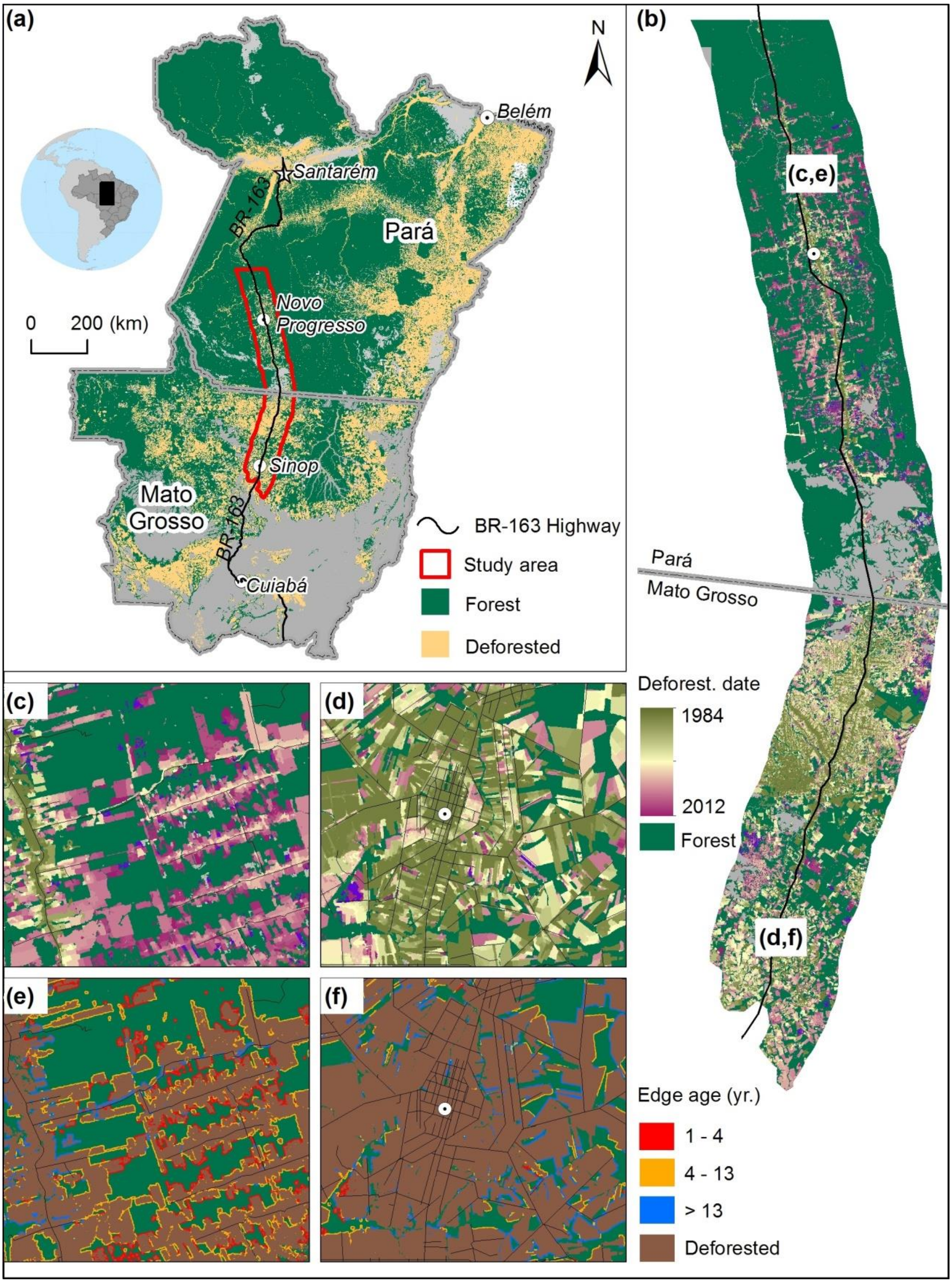
# 2. Material and methods

## 2.1. Study area

Our study area (Appendix Figure 1a) extends 50 kilometers around the BR-163 highway, following the road for 700km from the city of *Sinop* in Mato Grosso (south) beyond Novo Progresso in Pará (north). It covers an area of 81,648 $km^2$ from 5°30' to 12°54' southern latitude and 54°23' to 56°18' western longitude. The south-north natural vegetation gradient ranges from seasonal forests, over transitional evergreen-seasonal forests to open and dense evergreen forests (RADAMBRASIL 1975). Climate ranges from tropical wet and dry in the south (*Aw* in the Köppen Climate Classification) to tropical humid in the north (*Am* in the Köppen Climate Classification). Annual average temperatures range from 25.8 to 24.5 ºC and total annual precipitation from 1,800 to 2,300 (mm) in Sinop and Novo Progresso, respectively (AmbiWeb 2015). The dry season extends from June to September. The influence area of the BR-163 is in average drier than other areas in the Amazon Biome, which facilitates agricultural activities (agriculture and cattle ranching) in comparison with the surroundings (e.g., *Santarém*) (Fearnside 2007). On the other hand, the drier climate, combined with increasing fragmentation and forest degradation also might increase the region's susceptibility to forest fires.

The BR-163 highway is an important transportation corridor connecting the soybean producing areas in central Brazil with the regional and international marketplaces through the harbors of *Santarém* and *Itaituba* (Appendix Figure 1a). Deforestation began (Appendix Figure 1b-d) following a spontaneous and private colonization process triggered by the construction of the section connecting *Cuiabá* (Mato Grosso) to *Santarém* (Pará) in the early seventies (Coy and Klingler 2014).

Even though the Brazilian government intensified monitoring operations and implemented measures to halt deforestation and illegal timber extraction along the BR-163 (Fearnside 2007) and deforestation indeed has slowed down since 2004, the corridor is still a hotspot of forest loss (INPE 2014b). Today, the southern part of our study area, located in Mato Grosso, presents a stabilized agricultural landscape, dedicated to the production of soybeans, corn, cotton, and cattle ranching. The northern part in the state of Pará is an agricultural expansion frontier and the main economic activity is cattle ranching. With the road’s pavement in its final stage in the state of Pará, the BR-163 highway is likely to even become the main route for agricultural commodities transportation from the entire Mato Grosso state (Correa and Ramos 2010). The BR-163 paving will likely foster economic development by promoting the integration of regional markets, which in an insufficient environmental law enforcement scenario could further encourage the advance of the deforestation frontier towards the inner Amazon (Fearnside 2007).




**Appendix Figure 1. (a) Location of the study area (in red) in relation to the federal states of Para and Mato Grosso. (b) Annual deforestation in the study area (1984-2012).** Different patterns of deforestation around Novo Progresso/PA (c) and Sinop/MT (d). Temporal structure of detected edges for a recently deforested area around Novo Progresso/PA (e) and older deforested area around Sinop/MT (f). Sources: INPE (2014b); Müller et al. (2016a).

## 2.2. Datasets on deforestation, forest edges and biomass

We used old growth tropical forest cover and annual deforestation maps created by Müller et al. (2016a) covering the time period between 1984 and 2012 (Appendix Figure 1b-d). The authors used image compositing for 2,224 Landsat TM and ETM+ images across 11 footprints (Griffiths et al. 2013). They mapped stable forest in 1984 and identified subsequent deforestation events using a random forest classifier with an overall accuracy of 85% (Müller et al. 2016a). This dataset identifies deforestation on a per-pixel basis, i.e., at a 30 meters (m) spatial resolution, detecting clearing events as small as 0.1 ha. The authors stated that small patches of savanna vegetation in hilly terrain were misclassified as deforestation areas during the classification process. We therefore filtered deforested patches equal or smaller than 1 ha to avoid overestimation of deforestation and edge effect emissions associated with savanna vegetation. This approach decreased the overall deforested area by 3.7%. Based on a verification of 250 random samples we estimated that 82% of removed pixels were indeed associated with savanna vegetation, while 18% were true deforestation events. Additional non-forest area (water bodies, savanna vegetation and rocks) and edges of forest fragments neighboring natural non-forest land cover were excluded from the analysis using the forest mask provided by the Brazilian Amazon deforestation monitoring program (PRODES) data (see below) (INPE 2014b). This preprocessing ensured a conservative measure of forest edge creation.

We defined forest cover associated with forest edges for each year using buffers along yearly created deforestation patches (Numata et al. 2009). We introduced a buffer of 120 meters based on the findings of edge related biomass mortality by Laurance et al. (1998). Since there is little consensus about the extent to which degradation affects forest at edges (Broadbent et al. 2008; Chaplin-Kramer et al. 2015; Dantas de Paula et al. 2016) we tested an additional edge width of 300 meters (Numata et al. 2010; Shapiro et al. 2016). Edges that already existed at the starting year of our time series in 1984 were excluded from the analysis, as their age was unknown. From the resulting forest edge maps, we derived annual edge age information (Appendix Figure 1e-f) to account for biomass losses and carbon emissions from edge permanence.

To compare the benefits of long-term high spatial resolution data with PRODES (as a baseline that has been frequently used and is often referred to) we ran our analysis using PRODES as input. The PRODES assessment delivers annual spatially explicit information on deforestation for the Brazilian Amazon since the year 2000, with a declared minimum mapping unit of 6.25 ha. Therefore, cleared areas below this threshold are not necessarily detected. Recent literature has pointed that the importance of small clearings to total deforestation (6.25-50 ha) increased since 2002 (Rosa et al. 2012) with potential implications for LUCC carbon accounting. It is expected that deforested areas smaller than 6.25 ha have also proportionally increased. However, until now there were no datasets available to evaluate the importance of clearings smaller than 6.25 ha for deforestation and fragmentation carbon emissions. For this purpose, we compared both datasets to quantify emissions from deforestation and forest fragmentation since the year 2000.

To account for historical emissions, knowledge of original biomass is necessary. We therefore used a compilation of biomass content per vegetation type (IBGE 2004) based on Nogueira et al. (2008) and Leite et al. (2012). Nogueira et al. (2008) created a biomass map for the Brazilian Amazon combining inventory data and soil calibrated allometric equations applied to a forest type's map. Leite et al. (2012) compiled information for vegetation types not considered by Nogueira et al. (2008). The study area has a decreasing north-south biomass gradient, consistent with the Amazon-*Cerrado* - tropical forest to Brazilian Savanna - transition. Biomass ranged between 8 and 320t/ha. We used aboveground biomass as a proxy for landscape carbon stocks to estimate above and belowground carbon losses. We used a below/aboveground biomass ratio of 0.3 to compute the belowground root biomass pool (BGB) (Aguiar et al. 2012).

## 2.3. Modeling of carbon emissions

Our work is based on INPE-EM, a carbon book-keeping model proposed by Aguiar et al. (2012), similar to earlier frameworks (Houghton et al. 2000; Ramankutty et al. 2007). INPE-EM calculates both immediate and gradual greenhouse gases emissions on annual steps based on several parameters calibrated specifically to reflect forest removal techniques used in the Amazon. While INPE-EM models both fluxes from carbon sources (i.e., from old- and second growth forests removal) and sinks (i.e., sequestration from secondary vegetation regrowth), for this study we did not estimate the emissions balance. Instead, we calculate gradual carbon losses from deforestation and augmented INPE-EM to estimate carbon release due to edge effects. Originally, INPE-EM runs on the TerraME modeling environment (Carneiro et al. 2013), for the whole Legal Amazon using grid cells of 5 kilometers (km) cell size. For this study, we adapted INPE-EM to run on Dinamica-EGO (Soares-Filho et al. 2009) using the native Landsat data spatial resolution of 30 m. INPE-EM's original code is available for free download (inpe-em.ccst.inpe.br/).

Laurence et al. (1997; 1998) identified for a study conducted in Central Amazonia that 8.8% of the aboveground live biomass present in forest edges died during the first years after forest fragmentation and 1.8% was lost due to tree damage, totaling 10.6%, and that posterior losses were negligible. For modeling purposes, as in Numata et al. (2010; 2011) we distributed this potential biomass collapse (BC= 10.6%) linearly during the first four years following edge creation, assuming a linear rate of 2.65% $yr^{-1}$ (Equation 1). Subsequent to biomass collapse, carbon losses were booked linearly at a yearly rate of 10%. We considered that carbon content (C) represents 48% of the biomass and used a conversion factor (CF) of 3.67 to convert C to $CO_2$ (Aguiar et al. 2012). When edge deforestation occurred before the collapsed biomass carbon decay process completion the remaining carbon of the edge pool was accounted as deforestation emission, to avoid double booking. We considered that this remaining carbon content would be released through burning (Equation 2). Burned biomass emissions were divided into five greenhouse gases (GHG): carbon monoxide (CO), carbon dioxide ($CO_2$), nitrous oxide ($N_2O$), mono-nitrogen oxides ($NO_x$) and methane ($CH_4$) (Aguiar et al. 2012). We did not account for edge biomass recovery, since degradation favors shorter-lived successional trees and lianas, which have low wood density and biomass (Nascimento and Laurance 2004). The biomass map was updated annually by the edge component, meaning each pixel loses a percentage of original biomass according to edge permanence. As outputs we obtained annually updated biomass maps, which were used as input for the deforestation emissions module.

$$\text{Edge Permanence (EPe)} = (\text{Biomass x BC(Edge Age1-4}_{yr})) \text{ x } 0.1_{yr} \text{x C x CF} \quad \text{(eq. 1)}$$

$$\text{Edge Burning (EB)} = (\text{Edge Pool - EPe}) \text{ x GhG CF} \quad \text{(eq. 2)}$$

Annual deforestation data was overlaid with the biomass maps to estimate the yearly carbon release by deforestation. First, we considered that 15% of the aboveground biomass would be removed as commercial wood products ($P_{Wood}$ = 0.15). The 85% remaining aboveground biomass carbon content was divided between instantaneous emission through biomass burning ($P_{fire}$ = 0.42) and legacy emissions of two different carbon pools, i.e., slash ($P_{Slash}$= 0.41) and elemental carbon ($P_{ElemC}$ = 0.02) (Aguiar et al. 2012). Emissions from biomass burning were divided into five greenhouse gases, as explained above. Carbon release from wood products, slash and elemental carbon pools were distributed along the subsequent years following the deforestation event with different exponential decay rates (Equation 3). Due to the cyclic use of fire in agriculture, our model considers that the slash pool will re-burn (ReburnCycle) every three years, accelerating carbon release to the atmosphere

(Equation 3). Total emissions were calculated as the sum of carbon losses from edge effects and deforestation. We start the carbon accounting from the year 1985 since deforestation events detected in 1984 reflect deforestation that occurred in that year and in previous years.

$$\text{Deforestation} = ((P_{Wood})\text{x}0.1_{yr})+(((P_{Slash})\ \text{x}\ 0.4_{yr})\ \text{x}\ \mathit{ReburnCycle})+((P_{ElemC})\text{x}\ 0.001_{yr})+(\text{BGB x}\ 0.7_{yr}) \qquad \text{(eq. 3)}$$

We ran an additional model exploration using the dataset available from PRODES to assess the impact of using different deforestation data on estimated emissions between 2001 and 2012. To compare estimated emissions between the two products, we conducted the same procedures to create edge affected area and temporal structure using the data from PRODES. Appendix Supplementary Table 1 details data sources and parameters values for the different modeling explorations ran for this study.

### 2.4. Edge and fragmentation analysis

We analyzed three edge metrics to understand how landscape structure influences biomass stock changes and emission patterns in time and space. The edge-affected area is a measure of the absolute area of forest patch edges exposed, potentially causing forest disturbance (Numata et al. 2009). Large areas exposed to edge effects cause carbon losses; however, this process is only significant when the edge is not deforested quickly, allowing for the dead biomass to rapidly emit its carbon content. This process can be explained using the edge permanence time metric, which shows the average edge permanence time before it is deforested. The edge age composition was also calculated to illustrate how edges age and related carbon emissions decrease, especially if the landscape structure stabilizes (Numata et al. 2009).

## 3. Results

### 3.1. Deforestation and forest edge dynamics

In 2012, old growth forests covered 45,357 km$^2$, i.e., 38% less forest cover than in 1984. Between 1984 and 2012, 9,188 km$^2$ of forest were lost in Pará and 18,065 km$^2$ in Mato Grosso, 22% and 61% of the forest cover in 1984, respectively (27,253 km$^2$ overall). Yearly deforestation averaged 973 km² (1.7% yr), reaching a maximum cleared area of 2,172 km² in 2004. The PRODES dataset (2001-2012) identified more cleared areas than Müller et al. (2016a) (Appendix Supplementary Figure 1a-b) in the first years of the period, and less after 2006.

Accounted forest edge totaled 584 $km^2$ at the beginning of our monitoring period, 82% of which were in Mato Grosso. This area augmented to 5,727 $km^2$ in 2012, a tenfold increase. Pará experienced increasing edge affected area throughout the entire investigated period. Despite the expansion of deforestation in Mato Grosso, edge affected area stabilized after 2000 (Appendix Supplementary Figure 2) due to the balance between edge deforestation and edge creation (Numata et al. 2010), while net edge area increased along the active deforestation frontiers in Pará.

Edge age composition also explains the temporal patterns of edge biomass mortality and emissions. Only young forest edges (1-4 years) show active biomass mortality and from 2006-2012. Young forest edges accounted for 44% of all forest edges in the state of Pará, but only for 18% in Mato Grosso. Similarly, 5 to 13 years old edges, which are still emitting carbon were more present in Pará than in Mato Grosso (89% compared to 66%) (Appendix Supplementary Figure 3a-b). On average, 64% of all edges remained four years after creation in Mato Grosso, completing the biomass collapse process, and 29% remained for 13 years after edge creation, completing the carbon emission process. We identified less deforestation along edges in Pará, where; on average, 69% of edges remained four years after creation and 39% remained 13 years after edge creation (Appendix Supplementary Figure 4a-b).

## 3.2. Biomass and carbon losses from deforestation and edge effects

By 2012, 36% (974.3 Tg) of old growth forests biomass was removed, including belowground stocks. Of this total, 944.59 Tg (33.74 Tg $yr^{-1}$) of the biomass was lost via deforestation and 29.71 Tg (1.1 Tg $yr^{-1}$) due to forest fragmentation, representing 3.3% of the losses (i.e., deforestation and fragmentation related) (Appendix Table 1).

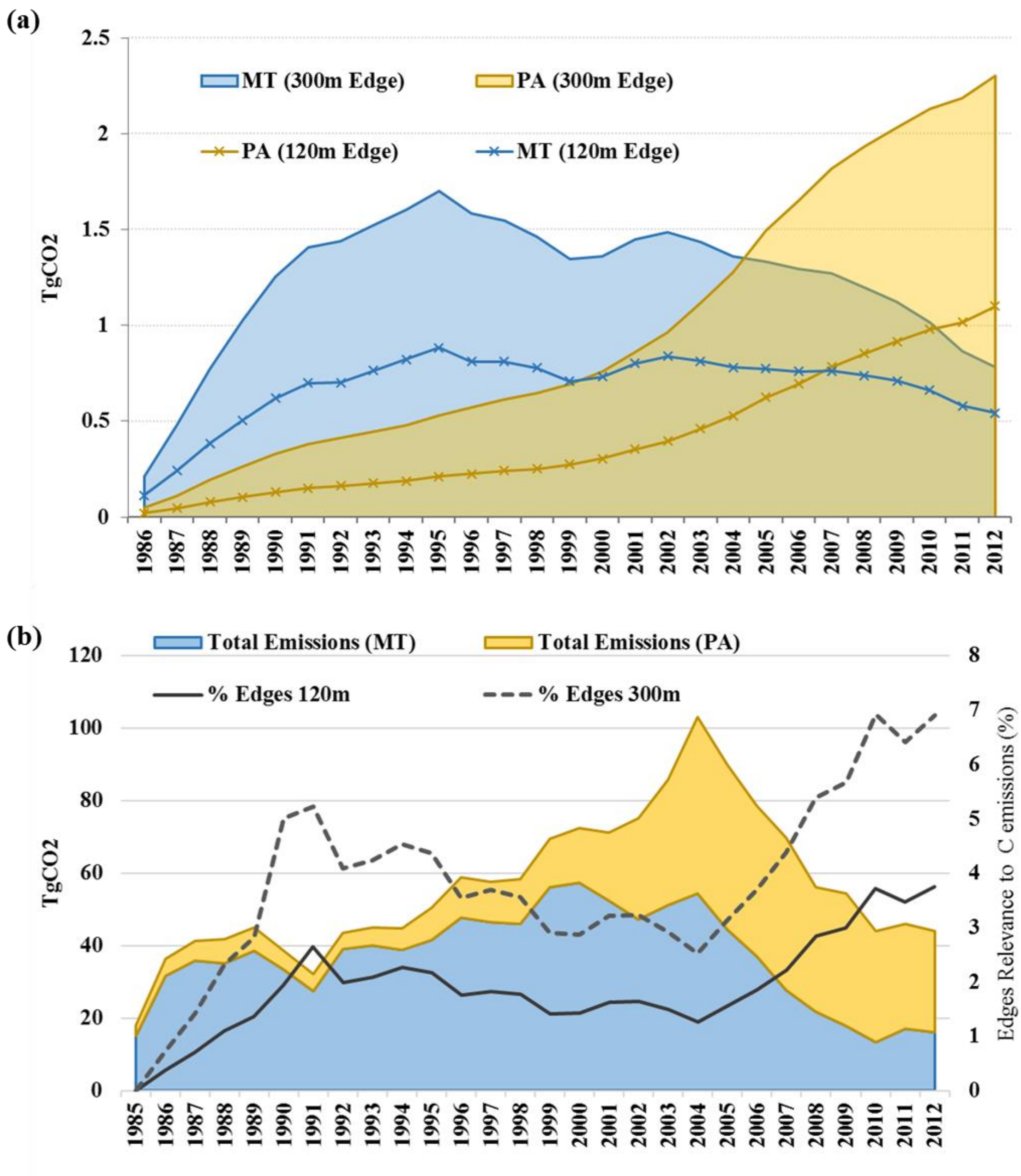
A total of 1,571.38 Tg $CO_2e$ was emitted between 1985 and 2012, from deforestation and edge biomass mortality, averaging 56.12 Tg $CO_2e$ $yr.^{-1}$ (Appendix Table 1). Carbon emissions increased until 2004, the year with that registered the largest forest loss, (Appendix Figure 2a) peaking at 103.13 Tg $CO_2e$. Mato Grosso accounted for 65% of total emissions, releasing 1,030.15 Tg $CO_2e$ by 2012 whereas Pará emitted 35% (541.22 Tg $CO_2e$) (Appendix Figure 2a). Important to notice, the area covering Mato Grosso represents 43% of our study area, which reinforces the importance of Mato Grosso on regional carbon losses. After 1999, the deforestation frontier advanced towards Pará, hence this state's emissions were concentrated in the second half of the investigated period (82%), whereas losses in Mato Grosso were distributed more evenly across the entire monitored period. After 2004, deforestation was in line with the generally decreasing trend observed for the Brazilian Amazon, with a lagged effect on carbon emissions.

**Appendix Table 1. Biomass losses and emissions from deforestation and forest fragmentation (1984-2012).**

| | Old Growth Forests Biomass Stock | Deforestation Biomass Loss (Tg) | Edge Biomass Loss (Tg) | Total Biomass Loss (Tg) | Deforestation Emissions (Tg $CO_2$e) | Edge Biomass Mortality Emissions (Tg $CO_2$e) | Fragmentation Relevance (%) | Total Emissions (Tg $CO_2$e) |
|---|---|---|---|---|---|---|---|---|
| **Total** | - | 944.59 | 29.71 | 974.30 | 1541.78 | 29.60 | 1.88 | 1571.38 |
| **1985** | 2519.79 | 31.76 | - | 31.76 | 18.07 | - | - | 18.07 |
| **Max (2004)** | 1801.92 | 76.48 | 1.38 | 77.86 | 101.82 | 1.3 | 1.26 | 103.13 |
| **2012** | 1577.24 | 16.98 | 1.17 | 18.15 | 42.23 | 1.64 | 3.74 | 43.88 |
| **Average** | - | 33.74 | 1.10 | 35.17 | 55.06 | 1.10 | 1.91 | 56.12 |

* Max = Higher losses for the time series

A total of 1,571.38 Tg $CO_2$e was emitted between 1985 and 2012, from deforestation and edge biomass mortality, averaging 56.12 Tg $CO_2$e yr.$^{-1}$ (Appendix Table 1). Carbon emissions increased until 2004, the year with the largest forest loss, (Appendix Figure 2a) peaking at 103.13 Tg $CO_2$e. Mato Grosso accounted for 65% of total emissions, releasing 1,030.15 Tg $CO_2$e by 2012 whereas Pará emitted the 35% (541.22 Tg $CO_2$e) (Appendix Figure 2a). Important to notice, the area covering Mato Grosso represents 43% of our study area, which reinforces the importance of Mato Grosso on regional carbon losses. After 1999, a new deforestation frontier began to expand in Pará, concentrating this state's emissions in the second half of the investigated period (82%), whereas losses in Mato Grosso were distributed more evenly across the entire monitored period. After 2004, deforestation was in line with the generally decreasing trend observed for the Brazilian Amazon, with a lagged effect on carbon emissions.




**Appendix Figure 2. Annual emissions from deforestation 1985-2012 (a) and forest fragmentation (b).**

Edge carbon emissions increased over the entire period (1985-2012) (Appendix Figure 2b). Edge emissions trends were less sensible to biomass losses than deforestation emissions (Appendix Figure 2a), since the edge carbon decay takes 13 years to be completed, whereas, in our model, most deforestation carbon losses are related to fast emitting pools (e.g., fire, BGB). Forest edge biomass mortality led to the emission of 29.6 Tg $CO_2e$ between 1985 and 2012 (0.36 Tg C $yr^{-1}$), with Mato Grosso accounting for 61% of edge losses. Edge emissions play a small role in the overall carbon losses (Appendix Figure 2a), averaging 1.88% during 1985-2012, and reached a maximum of 3.7% of the total emissions in 2010. The sensitivity analysis showed that considering a 300 meters' penetration of edge effects increases the emissions to 59.6 Tg $CO_2e$ (Appendix Figure 2a) in our study area (i.e., an additional emission of 30 Tg $CO_2e$ from edges), leading also to an increased average relative importance of edge-related emissions of 3.7%, peaking 6.9% in 2010, of overall carbon emissions. The reduced edge area creation combined with edge aging in Mato Grosso determined a downward trajectory of carbon losses triggered by forest fragmentation within the last 3 years of the time series (Appendix Figure 2a). On the other hand, the pronounced increase in edge affected area (Appendix Supplementary Figure 2) experienced in the 2000s boosted edge carbon losses in Pará (Appendix Figure 2a).

## 3.3. Deforestation, biomass losses and carbon emissions estimates using PRODES

According to PRODES, deforested area totaled 24,455 km² by 2012, of which 11,171 km² were cleared between 2001 and 2012, with an average rate of 931 km² $yr^{-1}$. For the same period (2001-2012), Müller et al. (2016a) detected a total deforestation of 12,559 km² at a rate of 1,046 km² $yr^{-1}$. Both datasets presented a marked decline of all clearing sizes contribution to total deforestation (Appendix Supplementary Figure 1a-b). One exception was the class size smaller than 6.25 ha, only detected by Müller et al. (2016a), which remained practically stable over time (Appendix Supplementary Figure 1a). Using the PRODES data, we detected a total of 558 km² of edges in 2001, increasing to 2,953 km² in 2012. This is 23 % less compared to the edge area calculated based on Müller et al. (2016a) for the same period (501 km² and 3,866 km², respectively).

The comparison between the PRODES and Müller et al. (2016a) datasets for the period between 2001 and 2012 revealed that biomass losses from deforestation and forest degradation were slightly smaller using the official PRODES product. Accumulated deforestation biomass losses totaled 415.32 Tg according to Müller et al. (2016a) and 371.61 Tg using the PRODES information (11% of difference). Regarding edge biomass mortality, the disparity was smaller as for Müller et al. (2016a) we estimated 12.58 Tg losses and using PRODES we obtained 11.52 Tg, 8.8 % less. Both datasets showed similar patterns of deforestation, despite annual variation in rates, with decreasing numbers after 2004. However, the reduction in deforestation rates detected by Müller et al. (2016a) was not as strong as detected by PRODES, which led to higher estimates using the first dataset.

The simulations from 2001-2012 using PRODES deforestation information yielded 9.4% lower overall carbon emissions compared to Müller et al. (2016a) (635.2 Tg $CO_2e$ and 701.5 Tg $CO_2e$). Since edge emissions have slower decay rates, the increased biomass losses detected using Müller et al. (2016a) as reference data were not translated into higher emission rates by 2012. Therefore, edge carbon losses estimated using the PRODES data as reference were 10.7% larger than using Müller et al. (2016a).

## 4. Discussion

In this paper we aimed for the quantification of carbon emissions from deforestation and forest degradation due to edge effects for the period between 1985-2012. Our findings showed that in 28 years our study area lost a considerable amount of old growth forest biomass (36% of stocks present in 1984) and consequently of carbon to the atmosphere (1571.38 Tg $CO_2e$). Moreover, 57% of biomass losses from deforestation and 54% of biomass losses from forest fragmentation occurred before 2001, - the year when spatially explicit information on deforestation became available from PRODES -, indicating the importance of long-term analysis for characterizing LULCC processes. On average, aboveground biomass density of deforested areas increased, due to the shift of forest loss from Mato Grosso to Pará, where higher-biomass forest types such as open and dense evergreen forests are more frequent. The increase of biomass density in deforested areas is in line with findings of other studies (Aguiar et al. 2012; Loarie et al. 2009; Ometto et al. 2014) and has implications for the overall magnitude of carbon losses.

Carbon losses along the BR-163 represented a significant share of overall deforestation driven emissions at state and biome levels. Using a similar biomass map and PRODES information, Fearnside et al. (2009) estimated for the whole state of Mato Grosso that, between 2006 and 2007, forest clearing removed 66 Tg $yr^{-1}$ of biomass, including the belowground pool. This means that for the same period, the BR-163 influence area accounted for 42% of this total, while at the same time representing less than 10% of the area. A comparison of our outputs with results from Aguiar et al. (2012) also shows the increasing relevance of the BR-163 corridor for the carbon losses in the Brazilian Amazon. Forest loss in the influence area of the BR-163 in relation to the Legal Amazon (BLA) increased from 6% in 2001 to 10% in 2012, reaching 14% of the BLA's deforestation in 2011. This increase was accompanied by a rise in the weight of carbon losses in the BR-163 when compared to the BLA of 7.9% to 9.9% for the same period. While it is yet unclear if post-deforestation management practices can enhance carbon soil storage (Boy et al. 2016) we show that deforestation itself is a relevant source of carbon, and that climate change prevention strategies should focus on avoiding future forest losses. Still, adequate land management holds great importance by restraining demand for additional land, improving livelihoods, having a positive impact on aboveground carbon stocks conservation.

This is the first study to map historical multi-decadal carbon emissions from deforestation along the BR-163 frontier. Deforestation rates varied significantly over the past three decades and across space (Appendix Figure 1a). Consistent with the frontier development along the BR-163, most of Mato Grosso's carbon losses occurred during the first 15 years of our analysis (1985-1999) while Pará presented higher losses from 2000-2012. By 2000, our study area in Mato Grosso was already a deforestation-saturated area, with most private properties presenting less forest area than the 80% legal reserve determined by the forest code (Stickler et al. 2013). In addition, between 1999 and 2001 the state government undertook a rural licensing program to control deforestation (Fearnside and Barbosa 2004). Consequently, the high opportunity costs of deforestation in Mato Grosso displaced the demand for land to southern Pará (Gollnow and Lakes 2014), increasing deforestation and carbon emissions after 2000. Short after, policy interventions (Assunção et al. 2015b; MMA 2012) and voluntary market mechanisms (Gibbs et al. 2015) decreased deforestation rates and related carbon emissions in both states. However, while the Brazilian Amazon deforestation rates have dropped to 70% below the 1996-2005 average (the period that Brazil uses as the official baseline), - bringing the country closer to the target of reaching the 80% reduction by 2020 -, in our study area the decrease was much lighter, of approximately 35% and only 18% in Pará. By shading light to the study area's historical trends, we were able to compare the region's baseline, which was unavailable before, to the Amazon's, making it clear that the BR-163 influence area is a source of LULCC carbon and a priority area for implementing law enforcement against illegal deforestation.

Next to Numata et al. (2010; 2011) this is one of the few studies that has quantified potential carbon losses related to edge effects for the Brazilian Amazon. Forest fragmentation represented a small share of biomass losses and carbon emissions throughout the investigated period, with an increasing importance after 2004 (Appendix Figure 2a). Numata et al. (2011) identified the same pattern when analyzing edge contribution to total emissions in the Brazilian Amazon. The relative importance of fragmentation carbon losses enlarged with pioneer frontiers consolidation. Edge biomass mortality, unlike deforestation, led to a continuous rise in carbon losses during 1985-2012 (Appendix Figure 2b), also contributing to a larger part of edge losses on total emissions, due to a few reasons. First, increasing forest losses expanded the edge affected area (Appendix Supplementary Figure 2), mainly in Pará, making more biomass vulnerable to disturbance and degradation. Thereafter, edge permanence time increased as deforestation rates dropped, making edges stable enough to complete the carbon decay process. However, the relative importance of edges to deforestation and fragmentation biomass losses in our study area is smaller than found by other studies, though. Numata et al. (2010) found that in the Brazilian state of Rondônia, edges accounted for 8.1% of combined biomass losses from forest clearing and edges, while we estimate a 2.95% contribution for our study area. This disparity is likely due to different fragmentation levels and deforestation geometry in the two regions, a pattern also identified by Laurance et al. (1998). In Rondônia, landscape is more fragmented and fishbone deforestation patterns are predominant, typical of the many rural settlements in the state, creating more edge affected area. Despite ongoing deforestation and fragmentation, our study area was still covered by large patches of forest and cleared areas followed a geometric deforestation pattern that is typical for medium to large size farms and leads to fewer edges and less biomass losses from edge effects. While deforestation reduction has been the target of many governmental policies, forest fragmentation has been a less prioritized issue in the Amazon. For instance, forest recovery requirements under the Brazilian Forest Code, combined with the creation of protected areas could consider landscape configuration in order to reduce carbon emissions from forest edge degradation.

The edge emissions model implemented was based on the work of Numata et al. (2010; 2011). However, although the framework was effective to assess fragmentation related carbon losses using extended time series of deforestation data, there are limitations to be addressed. Chief among those is the assumption that biomass mortality occurs at an identical rate in edges, at a fixed spatial range across the study area. Both parameters (mortality rate and edge width) are variable at the local scale depending on different environmental and ecological factors such as forest edge adjacencies to different land use types, extreme climatic events (e.g., droughts) and forest fires. For example, a study conducted in Central Amazonia by Mesquita et al. (1999) found that edges surrounded by pastures showed 55-100% higher tree mortality rates than edges surrounded by secondary forests. Edge effects also propagated further into inner forests surrounded by pastures (60-100 m) in comparison to forests surrounded by regrowing vegetation (0-60 m) (Mesquita et al. 1999). Episodic droughts also contribute to edge effects' variability. For instance, Laurance et al. (2001) investigated possible edge-effects associated with droughts caused by the El Nino-Southern Oscillation (ENSO) and found a significant increase in tree mortality rates during the ENSO drought, for a study area in Central Amazonia. Finally, fire occurrence could be relevant since most understory fires occur at up to 1 kilometer of forest edges (Berenguer et al. 2014). Future research should focus on a more detailed and dynamic representation of the heterogeneous nature of edge-effects in modeling carbon fluxes, including the above-mentioned processes.

In this study we undertook a sensitivity analysis to account for different ranges of edge effects and a potential underestimation of potential carbon emissions. Other studies have also pointed out that edge-effects penetration affects forests at a range larger than 120 meters, indicating that our study could have underestimated potential carbon emissions. Chaplin-Kramer et al. (2015) suggested that edge-effects could penetrate up to 5 kilometers inside forests. However, this figure might be an overestimation due to the coarser spatial resolution data (500 meters) used in the analysis (Chaplin-Kramer et al. 2015; Pelletier et al. 2013). Here, we test the effects of an edge width of 300 meters, which is in accordance with findings from recent studies (Shapiro et al. 2016) based on higher resolution remote sensing data. As expected, we found an increase in the relative importance of edges in relation to overall emissions, but lower than simulated by other studies (Chaplin-Kramer et al. 2015; Putz et al. 2014).

By comparing two datasets we could access data sources related uncertainty in carbon estimation. Both deforestation assessments used in this study identified periods of increase (2001-2004) and decrease (2005-2012) of forest loss, with impacting consequences for estimated carbon emissions. However, annual deforestation rates and annual carbon losses differed between the two datasets, with a variation larger than 20% for most years, even though both products are based on Landsat imagery. Small deforestation events not detected by PRODES (< 6.25 ha) partially explain the differences between datasets, accounting for 10% of biomass losses estimated based on Müller et al. (2016a). Small scale deforestation not only relates to PRODES limitations but also renders forest edges more important. In this context we caution an underestimation of carbon emissions provided by carbon estimates relying purely on PRODES data, especially for the period after 2004 (Rosa et al. 2012). Further causes for variation among datasets are related to underlying conceptual and methodological definitions. On the one hand, the PRODES assessment is based on visual interpretation using single date images chosen at the peak of the dry season, potentially leading to late detection depending on the acquisition date (Câmara et al. 2006). On the other hand, Müller et al. (2016a) use a compositing approach, which integrates all available cloud-free observations acquired during the dry season of a given year to extract the minimum tasseled cap wetness observation per pixel. This approach optimizes vegetation cover decrease detection and increases the chances of correctly labeling the occurrence year of deforestation events. In addition, low intensity deforestation processes are a widespread phenomenon in our study area (Pinheiro et al. 2016) where full canopy removal often takes 2–3 years to be completed. While low intensity deforestation is not explicitly included in the PRODES methodology, it leads to earlier deforestation detection in the approach by Müller et al. (2016a). These methodological differences lead to irregular mismatches between the products which are extremely difficult to track.

Uncertainties related to carbon accounting stem from data sources (i.e., deforestation and biomass datasets) and model parameterization. In this study we did not estimate the impact of different aboveground biomass datasets on biomass and carbon losses quantification. However, it is important to stress that a number of studies (Baccini et al. 2012; Nogueira et al. 2008; Saatchi et al. 2011; Saatchi et al. 2007; Sales et al. 2007) have provided aboveground biomass estimates for the Brazilian Amazon and tropical areas, showing considerable disagreement (Fearnside 2016; Mitchard et al. 2013). Recent spatially explicit biomass maps (Baccini et al. 2012; Saatchi et al. 2011) created from a combination of remote sensing products (optical and LIDAR) with field assessments represent a major step forward towards reliable biomass estimates. However, biomass estimates based on recently available remote sensing data have limited use for historic emission assessments.

## 5. Conclusions

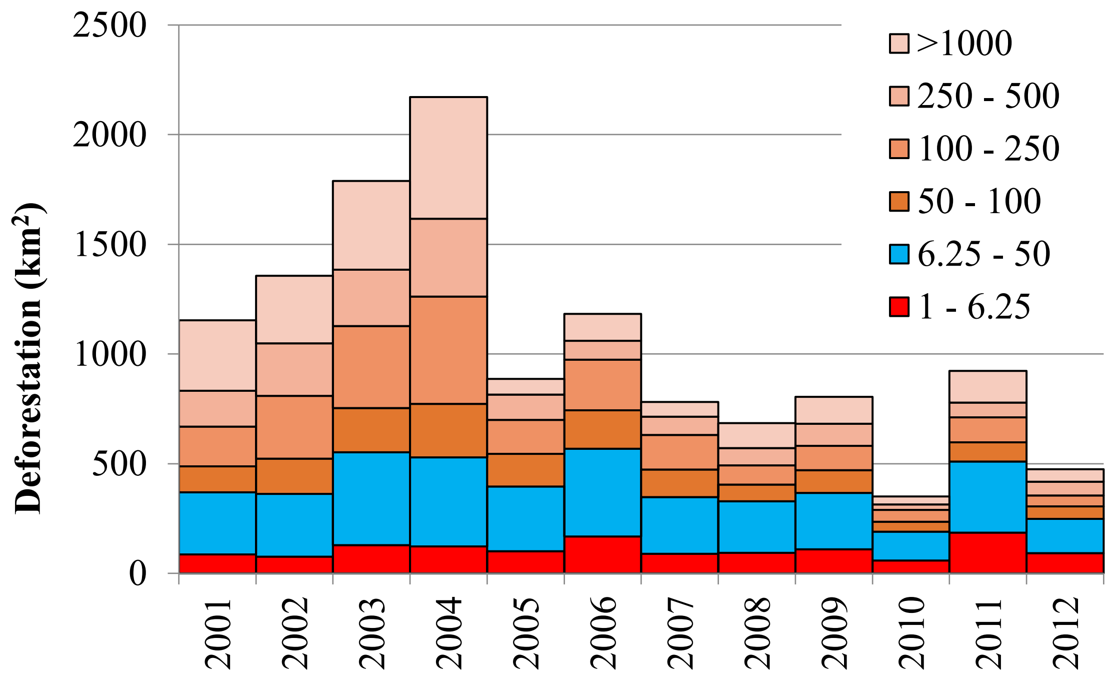
In this study we assessed the carbon implications of the expansion of a deforestation frontier in the Amazon. Our study profited from deforestation data dating back to the mid-1980s covering almost the entire development of the influence area of the BR-163 highway *Cuiabá-Santarém*. We conclude that this region followed the overall trends in deforestation and carbon emissions of the Amazon, but with a lesser decrease in forest losses, especially in Pará. This calls attention for policy makers about the need to direct more efforts to stop illegal deforestation in the region, and the potential of REDD+ programs. It is also important to mention that, besides uncovering past biomass losses and carbon emissions trends, extended time series of annual deforestation are fundamental for establishing carbon losses reduction targets baselines for REDD+ programs in sub national and regional levels.

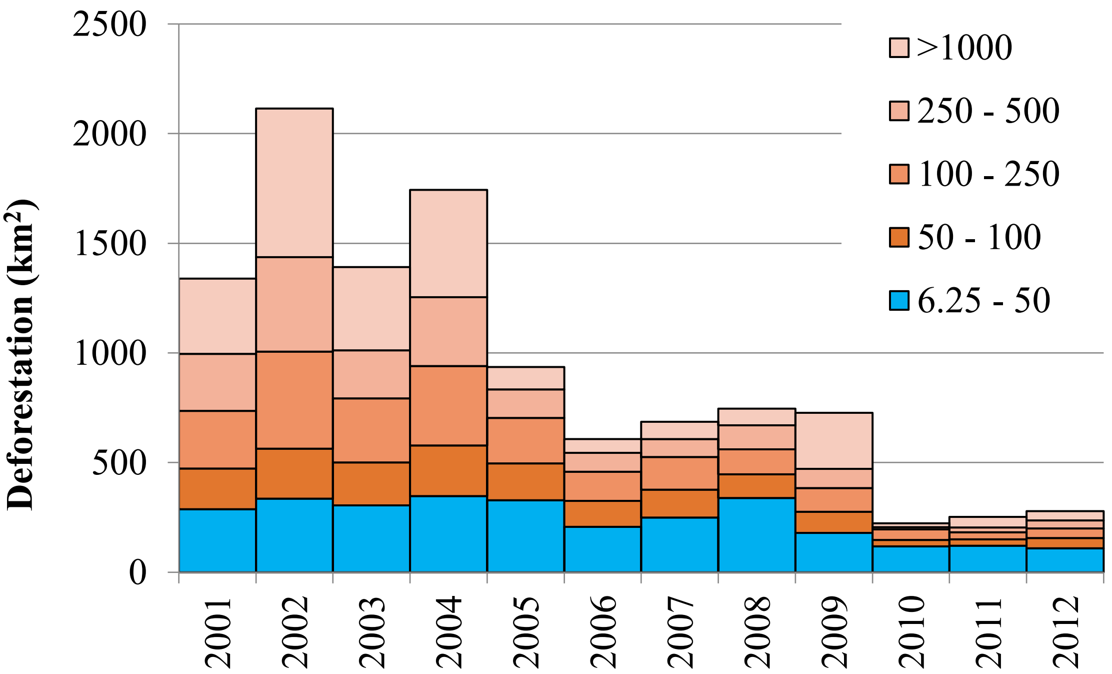
The extended time series allowed us to investigate long term potential carbon decay of fragmentation resulting forest edges, and to confirm the increasing relevance of edge effects on carbon emissions in consolidating deforestation frontiers. Next to Numata et al. (2010) this is the only study which provides such estimates based on a multi-decadal time deforestation series. However, it is important to stress that we offer an assessment of potential carbon losses, and that a thorough assessment of the geographical variation of edge effects intensity is a knowledge gap to be filled.

The high spatial resolution of the dataset, allowed us to investigate the contribution of small deforestation polygons (< 6.25 hectares) to carbon emissions, revealing a non-negligible contribution of small clearings to carbon losses. Furthermore, our approach improves carbon estimates reliability since it provides a quantification of uncertainty, caused by deforestation monitoring products. For that matter, we demonstrated that variations in deforestation products yielded average differences in carbon emissions of 11%, and PRODES based estimates likely underestimate carbon emissions since 2004.

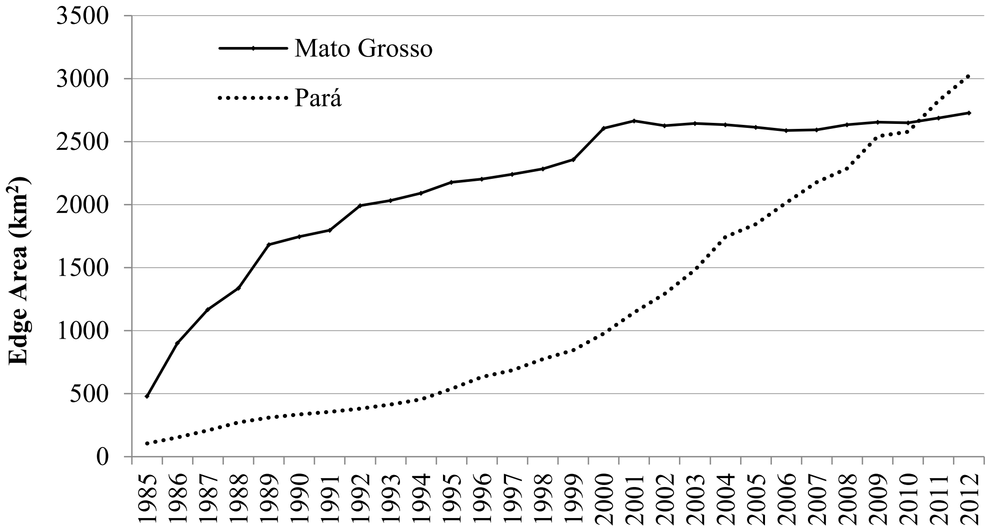
Whether deforestation and related carbon emissions will continue to drop or increase is uncertain, especially in southwestern Pará, a region particularly vulnerable to speculative deforestation (Fearnside 2008; Margules 2004). Recent reports by PRODES indicated spikes in deforestation rates in 2013 and 2015 after one decade of sharpen decrease, a red alert for policy makers that the current deforestation prevention model might be exhausted and that new approaches could be necessary. Therefore, the future of carbon stocks from the Amazon is highly uncertain and more detailed studies, e.g., confronting monitoring products uncertainties and limitations and including an improved representation of edge-effects and forest changes in modeling carbon fluxes, will be needed in the future to support decision making.

# 6. Supplementary material

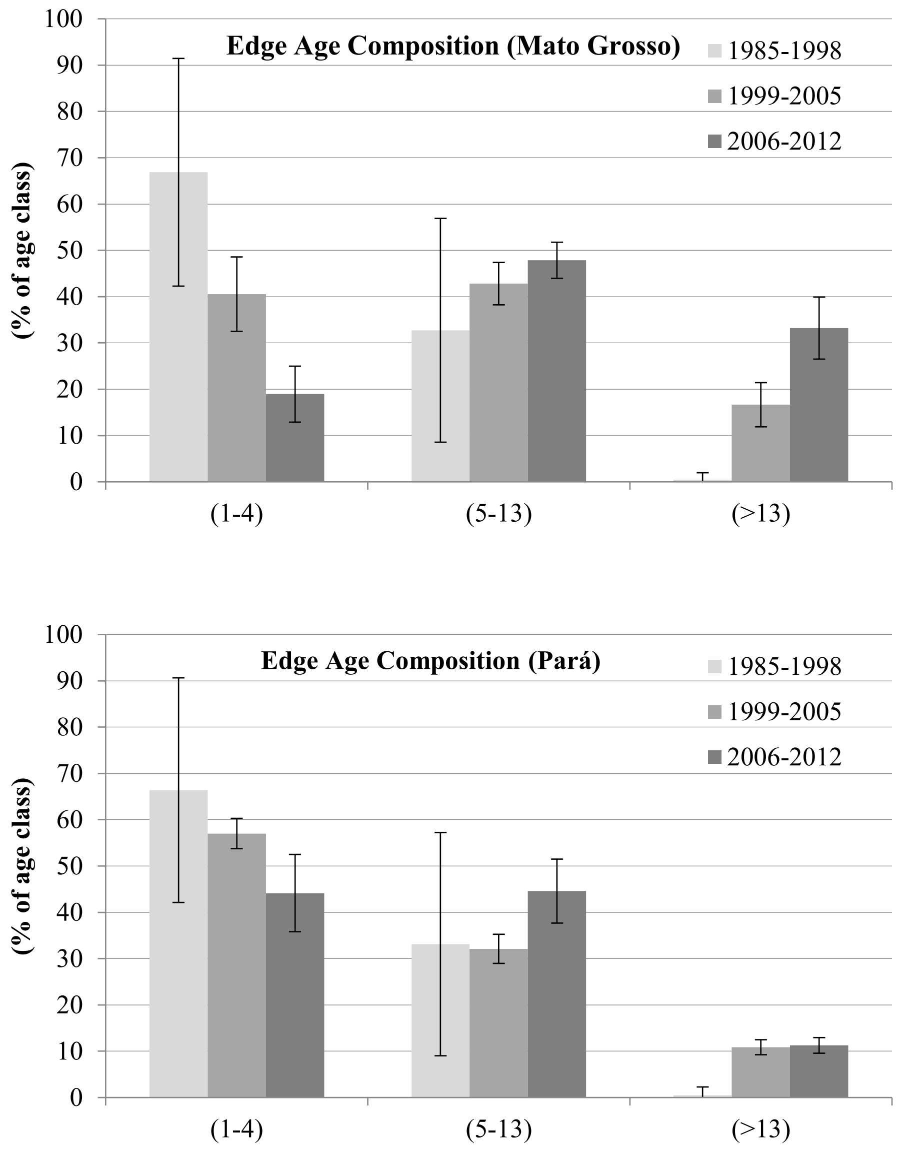


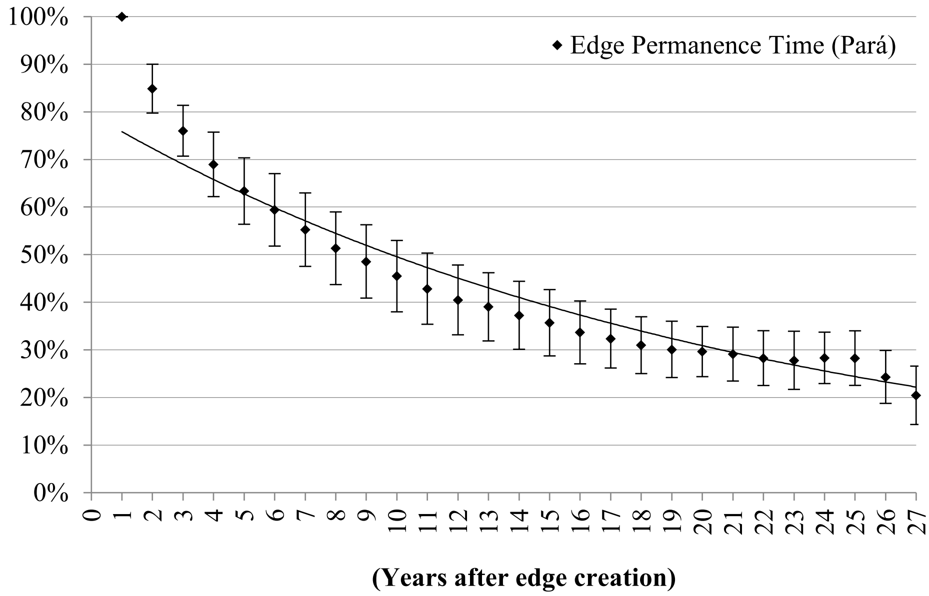

**Appendix Supplementary Figure 1. Annual deforestation (square kilometers) stratified by cleared patch size (hectares) (a) Müller et al. 2016 and (b) PRODES.**

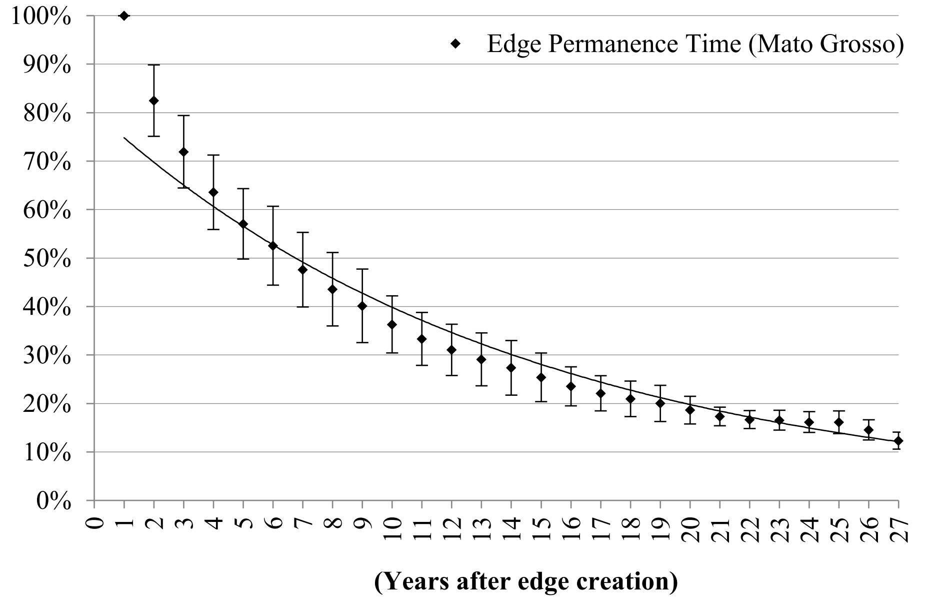



**Appendix Supplementary Figure 2. Edge area evolution in Mato Grosso and Pará states (1985-2012)**




**Appendix Supplementary Figure 3. Average edge age composition. (a) Mato Grosso (b) Pará;**




**Appendix Supplementary Figure 4. Edges permanence time** - (a) Mato Grosso and (b) Pará (Bars represent standard deviation for the time series)

**Appendix Supplementary Table 1. Carbon bookkeeping model parameters and combinations used in this study.**

| | | Model Run – Müller et al. 2016 Dataset | | | | | | Model Run – PRODES Dataset | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | *Edges* | | | | | | *Edges* | | | | |
| | | Baseline | Sensitivity Analysis | Spatial Dimension | Temporal Coverage | Unit | Source | Value | Spatial Dimension | Temporal Coverage | Unit | Source |
| Edge module parameters | Potentially affected edges | 120 | 300 | Spatially Explicit | (1985-2012) | meters | Müller et al. 2016 Numata et al. 2010 Shapiro et al. 2016 | 120 | Spatially Explicit | (2001-2012) | meters | PRODES 2012 |
| | Edge biomass mortality | 2.65 | 2.65 | Non-spatial | | % | Laurance et al. 1997; Numata et al. 2010 | 2.65 | Non-spatial | | % | Laurance et al. 1997; Numata et al. 2010 |
| | Decay rate edge biomass | 0.1 | 0.1 | Non-spatial | | n.a. | Numata et al. 2010 | 0.1 | Non-spatial | | n.a. | Numata et al. 2010 |
| | | | *Deforestation* | | | | | *Deforestation* | | | | |
| Clear-Cut Deforestation module parameters | Deforestation | ≥1 | ≥1 | Spatially Explicit | (1985-2012) | hectare | Müller et al. 2016 | ≥ 6.25 | Spatially Explicit | (2001-2012) | hectare | PRODES 2012 |
| | Slash fraction | 41 | 41 | Non-spatial | | % | Aguiar et al. 2012 | 41 | Non-spatial | | % | Aguiar et al. 2012 |
| | Wood Products | 15 | 15 | | | % | | 15 | | | % | |
| | Burned first year | 42 | 42 | | | % | | 42 | | | % | |
| | Elemental carbon fraction | 2 | 2 | | | % | | 2 | | | % | |
| | Reburn cycle | 3 | 3 | | | years | | 3 | | | years | |
| | Decay rate wood | 0.1 | 0.1 | | | n.a. | | 0.1 | | | n.a. | |
| | Decay rate slash | 0.4 | 0.4 | | | n.a. | | 0.4 | | | n.a. | |
| | Decay rate elemental carbon | 0.001 | 0.001 | | | n.a. | | 0.001 | | | n.a. | |
| | Decay rate BGB | 0.7 | 0.7 | | | n.a. | | 0.7 | | | n.a. | |
| | | | *Biomass* | | | | | *Biomass* | | | | |
| Biomass data | | 8-320 | 8-320 | Spatially Explicit | (1985-2012) | Ton/ha | Leite et al 2012 | 8-320 | Spatially Explicit | (1985-2012) | Ton/ha | Leite et al 2012 |

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# Eidesstattliche Erklärung

Ich erkläre, dass ich die Dissertation selbständig und nur unter Verwendung der von mir angegebenen Hilfsmittel (gemäß §7 Abs. 3 der Promotionsordnung der Mathematisch-Naturwissenschaftlichen Fakultät, veröffentlicht im Amtlichen Mitteilungsblatt der Humboldt-Universität zu Berlin Nr. 42/2018 am 11.07.2018) angefertigt habe. Ich habe mich nicht anderweitig um einem Doktorgrad im Promotionsfach Geographie beworben und besitze keinen Doktorgrad im Promotionsfach Geographie. Die Promotionsordnung der Mathematisch-Naturwissenschaftlichen Fakultät, veröffentlicht im Amtlichen Mitteilungsblatt der Humboldt-Universität zu Berlin Nr. 42/2018 am 11.07.2018 habe ich zur Kenntnis genommen.

Letícia de Barros Viana Hissa

Berlin, den 26.11.2019